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Proceedings—Management and Productivity of Western-Montane Forest Soils

Boise, ID, April 10-12, 1990

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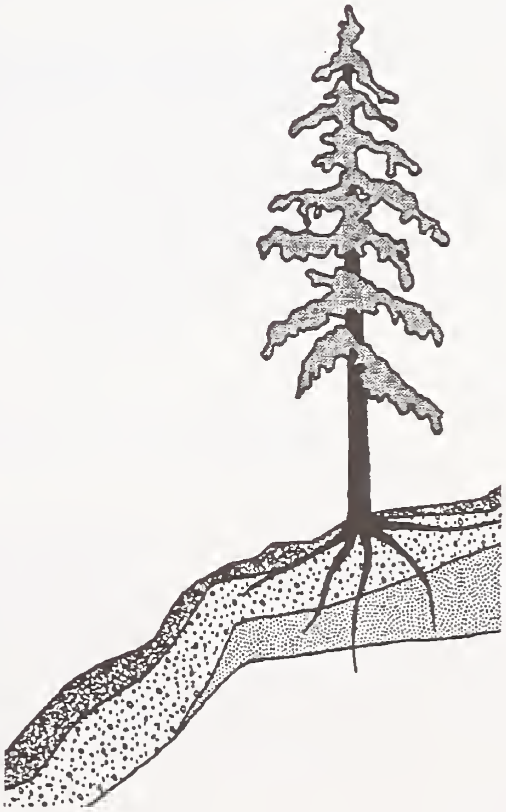

FOREWORD

The reason we held the western-montane forest soils symposium was simple: a concerted professional response to a developing informational crisis! We were in need of a "state-of-the-art" publication on the nature and problems of integrating soils information and expertise into the management of inland western ("western-montane") forest resources. Soils of the region are perceived to be largely fragile and easily damaged. Relatively little information has been generated locally and many of the resident soils experts are near or past retirement. Thus, our intention was to emphasize regionally available material, to incorporate pertinent information from the world literature, and to capture otherwise unpublished wisdom from retiring regional experts.

As important as soils are to forest health and productivity, sources for the information are declining as the demand increases. Soil management has become information intensive. Reasons include rapidly increasing numbers of site-specific forest plans throughout the Nation, global climate issues, acid rain issues, long-term forest productivity and health issues, demands for clean water, new technology (primarily GIS mapping of forest and soil productivity), and a growing demand from international research communities.

If the symposium has been successful in making applicable information more available and useful for professional managers in this highly sensitive region, it will provide a worthwhile contribution.

As always in such an undertaking, accomplishments are made through efforts of many. Specific recognition is accorded to Carol Spain of the University of Idaho and Jonalea Tonn of the Intermountain Research Station. They did outstanding work in handling logistics to develop the agenda and make the meeting run smoothly.

The organizing committee for the symposium consisted of Leon F. Neuenschwander, professor of forest resources, adjunct associate professor of range resources, associate dean for research and international programs, and associate director of the Idaho Forest, Wildlife and Range Experiment Station; Alan E. Harvey, project leader for the Intermountain Station's Forest Diseases and Microbiology of the Northern Rocky Mountains research work unit in Moscow, ID; Carol Spain, coordinator for Employment Services and Continuing Education and Outreach; and Jonalea R. Tonn, forester, Intermountain Research Station in Moscow, ID.

The outstanding subject matter covered by the agenda was a direct result of deliberations by an exceptional technical committee. This committee included David Breuer, then of Washington State University; Jim Clayton, Intermountain Research Station (Boise); Richard Fisher, Utah State University; Gary Ford, Idaho Panhandle National Forests; Mike Geist, Pacific Northwest Research Station (La Grande); Russ Graham, Intermountain Research Station (Moscow); Alan Harvey, Intermountain Research Station (Moscow); Glen Klock, representing Western Resources Analysis (Wenatchee, WA); Martin Jurgensen, Michigan Technological University; Geral McDonald, Intermountain Research Station (Moscow); Dale McGreer, Potlatch Corporation; Bob McDole and Leon Neuenschwander, University of Idaho; Jerry Neihoff, Idaho Panhandle National Forests; Debbie Page-Dumroese, Intermountain Research Station (Moscow); Jonalea Tonn, Intermountain Research Station (Moscow); and Dale Wilson, Clearwater National Forest.

We all hope that users of the information in this proceedings will find it helpful in dealing with the unique qualities, potentials, and liabilities inherent to western-montane forest soils!

—Al Harvey
Lee Neuenschwander

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FOREST PRODUCTIVITY: LESSONS TO BE LEARNED

Richard G. Cline
Nelson S. Loftus

ABSTRACT

This paper is a discussion of our research progress in increasing forest productivity relative to that in cropland agriculture. Key differences seem to be due to the intense cooperation among various agronomic specialties, their ability to measure product yields over short periods of time, and the availability of data sets containing very specific crop / species, soil, climate, and fertility information. The lessons to be learned from field-crop production research include the value of ecological concepts, a focus on basic processes and controlling factors, the value of observation, the need to be active rather than reactive, and the advantages of working together.

INTRODUCTION

Forest productivity seems at times to be looked upon as a special, and somehow unrelated, aspect of agriculture. This view seems a bit narrow. Forestry, after all, is based on essentially the same biological processes as the rest of agriculture. This being true, what can we do to take advantage of the relationships?

First, let us assume agricultural production (including forestry) is based primarily on our ability to retrieve a plant or plant-derived product from our environment. What we use it for after that is a related but secondary question. We could use it to feed ourselves, to feed animals that we subsequently eat, to produce fiber that we use, or to produce a recreational environment to be enjoyed. All of these things have value. We harvest them all in one form or another.

For the sake of argument and comparison let us look at plant products that can be produced on the land and sold after harvest. Let us compare productivity, and our understanding of it, in cropland agriculture with that in forestry. We have a long history of production in both. Our ability to achieve production improvements, however, has been much more impressive in our croplands than in forestry. Why?

The first general thought that comes to mind is historic. We have a rich history of intense cooperation among the various agronomic specialties in cropland agriculture. This has produced some truly impressive synergisms, particularly among the fields of crop science, soil science, and genetics. The yield of wheat, for example, has more than quadrupled in the past 50 years. This cooperation is one of the main reasons for our rapid increase in crop production during the last half century. This kind of cooperation has often been nearly absent from the field of forestry. Why? Are the growth processes somehow different in forests than in field crops? That does not seem very reasonable. All plants operate pretty much the same way within certain limits. The same basic principles should apply if one allows for species and culture differences.

COMPARING MEASUREMENT

Maybe we can shed some light on this problem by comparing production measurement in these two areas of agriculture. Table 1 attempts to compare the process of production measurement in croplands with that of forests. While the comparison is not absolutely a true one for all cases, it seems correct in a general sense. The table suggests many more similarities than differences. This seems logical. The differences, while apparently few, are glaring, at least in practice.

Table 1— A comparison of productivity measurement in cropland and forest agriculture

Measurement considerations	Cropland	Forestland
Measure	Harvested product (grain)	Harvested product (fiber)
When measured	Yearly	Over the rotation
What measured	Mass (weight) Volume (when sold)	Mass (weight) Volume (when sold)
Measurement method	Scales (grain elevator)	Scales Volume measurement Volume estimates
Measurement context	Species, often variety specific	Often not species specific
	Location specific (known soil, fertility levels)	Not location, soil, or fertility-level specific
	Cost vs. return usually well known	Cost vs. return often difficult to determine because of rotation length

Paper presented at the Symposium on Management and Productivity of Western-Montane Forest Soils, Boise, ID, April 10-12, 1990.

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The harvested product seems like an obvious choice for a first view, and should provide a good basis for evaluation, providing good measures of the harvested product can be obtained. Similarities are much more obvious than differences here. The period of measurement is a bit more of a problem. Field crops are usually harvested during the same growing season they are sown, and reasonably reliable measurements can be obtained at that time. Forest products, trees for this example, are harvested over a rotation that can be 80 to 100+ years. At other times, field measurements and estimates with varying degrees of reliability are used. Harvests can range from partial cuts through thinnings, where residual volumes are estimated, to clearcuts where all trees are removed and merchantable or total volumes are estimated. This is an important difference even though the units of measure might be quite similar—for example, tons of grain and tons of green wood. Another major difference is the measurement context. It seems the context of measurements in forestry is usually much less specific, making it more difficult to obtain useful information.

A summary of this comparison of cropland and forest agriculture makes three points:

- We in forest science seem not to have brought the full range of our scientific weapons to bear on the productivity problem as has happened in cropland agriculture. We tend not to work together. The reasons are not important. We need to do it. The potential progress and benefits can be significant if the results in other areas of agriculture are any indication.

- Our inability to measure our product reliably at short intervals hampers progress. This seems like a major problem. The long rotation length of forest crops makes it much more difficult to accumulate the largely empirical relational data sets critical to developing working mechanistic hypotheses. These hypotheses and the research that they generate provide the specific understanding needed to take advantage of the array of scientific synergisms seen in field-crop production.

- Our lack of data sets containing sufficient species and growth environment (soil, climate, fertility) specificity is another problem. The problem seems different, but its effects are essentially the same as those related to long rotations. We do not have the empirical data so necessary to developing our concepts of mechanistic and process relationships.

Research to determine productivity or response to treatment has historically depended on field-plot-based experiments. Results of this work have been used to infer the effects of management inputs on outputs given a defined set of resource properties. Our history in the more extensively managed forest environments has been less successful than that in croplands and more dependent on axiomatic information. Our efforts have been hindered by the difficulty of adequately controlling experiments in a highly variable wildland environment. This is not to suggest that we have no good research results or that there is no empirical data base to work from. We do have data, and good research has been done.

Obviously the returns for developing this kind of knowledge and understanding for our intensively managed croplands are much greater than they are for our forestlands. It is logical, then, to expect more effort in those areas where returns are greater. This is likely to continue in spite of increasing public attention on and demands for resource returns from forested wildland environments. We will need better data and firmer understanding of these environments to form the inferential foundations for future management decisions as these demands increase.

WHAT CAN WE DO?

It seems we have a challenge before us. We must overcome a limited, by comparison, availability of research data, probably research funding, expertise, and the continuity of that expertise in the future. We must be more efficient—use the data we have more efficiently and creatively and develop a more integrated understanding of our resource base. There are a number of specific things we can do. They are not new. They have been suggested before, but we probably have not made adequate use of them. Maybe, fuddy duddies that we are, we have trouble forcing ourselves out of the rut in which we happen to find ourselves. Here is a short list of some things to try, or try again:

- Use ecological concepts to establish reference points and a basis for data extrapolation and interpretation. We have tried this before, often with considerable success. Habitat type classifications and their growing utility are but one example. There are a variety of other possibilities. We also need to recognize, however, that classifications are all artificial constructs. They are tremendously valuable as a communication tool and as a method of extrapolating inference through inductive reasoning. The fact remains, however, that being artificial constructs of our momentary and ever-changing concepts, they need to be constantly challenged, tested, and revised. Their value is that they are capable of helping us organize our thoughts about complex landscape-based ecosystems, isolate the properties of those systems most useful to us, and facilitate our efforts to overcome our lack of adequate empirical data.

- We need to focus on the basic processes and factors controlling productivity. This is actually a corollary of the previous point because the ecological concepts, if correctly conceived, will be related to those basic factors and processes. Our ability to correctly identify these things will be considerably enhanced by forming the alliances, previously mentioned, that have been so successful in cropland agriculture. Having increased our productive capacity for food, why should we ignore the potential for doing the same for forested environments, whether we are producing fiber, wildlife, or scenery?

- We need to relearn the value of observation. There is a strong tendency in the scientific literature to discount anything that does not have a number or a statistical procedure attached to it. Research should never be undertaken without adequate statistical preparation, but

regression, for example, does not prove cause and effect. Some of the most useful literature we have seen has been among the older articles where conclusions were more dependent on observation and where the logic connecting observation and conclusions seems better developed than is routine with our more sophisticated statistical procedures. Statistics are not a substitute for understanding what is observed of nature.

•We need to resist the temptation to allow crisis to drive programs. Too often, we are driven by short-term objectives. This tendency is not going to go away. We will all succumb to the short-term imperative at times, but we must make sure, through it all, that we maintain (reestablish where necessary) the utility and value of long-term data. This is particularly important when the crop we grow is a long-term crop. If the long-term experiments of Rothamsted and other places are valuable for short-rotation species, think how valuable they must be for species with much longer rotations.

•We need to take a lesson from our colleagues in cropland agriculture. We need to take advantage of the combined experience of the variety of specialty areas within both forestry and agronomic sciences. We need to work together more than we do now.

The discussion, to this point, has concentrated on commodity production by plants. Forests have much more to offer. Recreation opportunity, esthetics, and high-quality water are but three examples. The demand for these resources, and their value, seems to be increasing. This trend is expected to continue. We need to develop better productivity relationships for these resources because we are being asked to produce them. Our management systems need to provide reliable estimates of our ability to produce a complex array of values.

Speakers answered questions after their presentations. Following are the questions and answers on this topic:

Q. (referred by Glen Klock)—About 95 percent of the talks in this "Forest Soils Conference" have not used any reference to soil taxonomic units. Are we really using soils as currently defined in a taxonomic system, or do we get enough soil direction by characterizing such factors as soil compaction, soil erosion, nitrogen and carbon contents, and mycorrhizal conditions? If generalizations are used, how do we extrapolate data and communicate with others? Are we really preaching to the converted?

A. (from Cline)—I will answer the last part of the question first because it is the easiest. Yes, we probably are preaching to the converted, especially at a conference like this attended mostly by resource professionals and most of those probably soil scientists. It is unlikely to be otherwise in this situation. The value of the conference, of course, is not the preaching. It is the exchange of ideas and the publication we can use to inform others.

Concerning the apparent lack of taxonomic class reference in papers presented, I would suggest that taxonomic classes are only useful if their specific interpretive value is known and well documented in relation to the soil properties we have been discussing. I would agree that the taxonomy has extrapolative value. It is also an excellent vehicle for communicating commonly understood ideas. It is, as are all classifications, a substantially artificial system reflective of the thoughts of its framers at the time it was constructed. You might be aware that we are continuing to change it as we learn more about soils.

Another point concerning the soil taxonomy seems important. It is, on the whole, a general system designed to reflect concepts of the whole population of soils as we presently understand them. This seems to be its main objective. We need that kind of an overall view to help us define and communicate ideas in common. Mapping problems and their associated local interpretations require a different objective, consequently a different classification design. It is basic to the logic of classification systems that, for optimum utility, the system design should change when its objectives change.

I will provide two examples using taxonomy as a reference to make this point. The taxonomy uses 35 percent rock fragments as the class break between skeletal and nonskeletal families. Remember that this must be imposed downward on the series, taken as a taxonomic class. The field criteria for map unit design in a survey with which I am familiar placed the class break for map units near 20 percent. This suggests that the practical interpretive criteria did not fit the taxonomic criteria very well. The solution was to ignore the taxonomic criteria for survey purposes and handle the two taxonomic classes involved (20-35 percent and 35 to something over 55 percent) as similar soils.

Another instance of map unit design at variance with taxonomic criteria relates to classes of Andepts (35 cm of andic, volcanic ash, material at the surface) and andic subgroups (18-35 cm of andic material). This material has some unique and valuable water-retention characteristics. These soils on steep south aspects seemed to need about 25 cm of this material to support certain types of productive timber stands and provide assurance of stand regeneration. The map unit design described Andepts and andic subgroups with ash caps greater than 25 cm as similar soils in some map units and andic subgroups with typic subgroups (less than 18 cm) as similar soils in others. The same kinds of soils on steep north aspects in the same survey were not separated at the same boundary (25 cm ash cap), because the water relations were not the same.

The point of all this is that, with changing class separation objectives, class boundaries need to change. This is just fine for local interpretive purposes. The more general, formal taxonomies provide us with our conceptual and communication tools. They need the stability associated with consistently established class definitions. We need to recognize the value each has in our scheme of things and avoid making ourselves slaves to either. We build map units and interpretations from the soil properties and associated biological and physical responses we observe. We develop concepts and communicate using consistently defined sets of properties that we understand in common.

SOILS INFORMATION: A VALUABLE TOOL FOR MANAGEMENT OF WESTERN-MONTANE FORESTS

Richard F. Fisher

ABSTRACT

Soils information mapped and interpreted at large scales can be very useful in forest management. Such information is objective rather than subjective. It can be aggregated for use at smaller scales and can be placed free of scale errors into geographic information systems. It can be interpreted for a wide variety of uses. Unfortunately, our ability to collect and interpret such information for western-montane forests in a timely manner is severely limited by a lack of trained personnel and appropriate basic studies.

INTRODUCTION

To say that vegetation is not related to the soil on which it grows is akin to saying that vegetation is not related to the local climate. Yet we not only hear this statement said, but we also see it written. If we compare 1:500,000 maps of Koppen's climatic regions with similar scale maps of biomes they compare favorably, but if we were to compare the efficacy of Koppen's system for predicting vegetation types at a scale of 1:12,000, we would be forced to conclude that vegetation and climate are not related.

Similarly, Marbut's soil great groups match biomes fairly well in eastern North America. They fail miserably in the West due to the rapid changes in both soils and vegetation with changes in altitude. Certainly if we investigate the efficacy of Marbut's soil units for predicting vegetation type at a scale of 1:12,000, we must conclude that vegetation and soils are not related.

Of course we know that vegetation is related to microclimate, and if we measure and map microclimate we can predict vegetation types fairly well even at a scale of 1:12,000. Soil should have similar efficacy, microsoil that is. Unfortunately, we have not looked much at soil in this detail in the West. In the Southeast, where very small changes in elevation (<5 ft) lead to large changes in both soil properties and tree growth, we have looked at soils in great detail. This investigation has revealed that not only vegetation type but even growth is closely related to soil type (Fisher 1984).

In the Mountain West, due to the cooler or drier environment and the youthful landscape, large changes in elevation (>100 ft) may yield only minor changes in vegetation and soil. This has certainly helped to foster the idea that soil does not drive vegetation in the West, but there are many other factors that contribute to soils' poor image.

Paper presented at the Symposium on Management and Productivity of Western-Montane Forest Soils, Boise, ID, April 10-12, 1990.

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INFORMATION DEMAND

When lands are managed extensively, the demand for information to aid management is small. As we intensify management or increase investment in land, then the demand for information climbs sharply (fig. 1). By-and-large, forest management in the Mountain West still falls into the exploitation part of this graph. As we in the West manage fewer acres more intensively for wood production, recreation, etc., we will move to the left, and our informational needs will rise exponentially.

We will begin to worry about stands at a scale of 1:20,000 or larger rather than at 1:50,000 or smaller. At those scales we can collect valuable soils information and utilize it effectively. At those scales land types and habitat types become less useful due to their breadth. Also, the increased value of information means that we can expend more money and energy to obtain the information.

This brings us to the second reason why soil has received so little attention in the Mountain West. Soil-vegetation relationships, although theoretically simple, are in reality very complex. There are many indirect as well as direct relationships and many feedback loops. We must look at soils very carefully if we are to develop models that predict vegetation type, let alone growth. Few such indepth studies have been carried out in the Mountain West.

RESEARCH TO DATE

Much of what has been done has been superficial or empirical. Habitat typers have assumed that geology is more important than soils, and that may be correct at the high level of generalization at which they work. When they have looked at soil it has only been at surface soil, which is seldom a good predictor of anything (Pfister and others 1977; Pritchett and Fisher 1987).

Some workers have attempted to develop empirical relationships between soil properties and vegetation or growth of vegetation. This has been done through a multiple regression approach, which has many pitfalls. There are other analytical approaches such as discriminate analysis and classification trees, which may be more appropriate, although they remain empirical (Verbyla 1986; Verbyla and Fisher 1989).

Unfortunately, we have developed little true theory in the area of soil-vegetation or soil-site analysis. Not that theory in and of itself is good. Empiricism advances our ability to predict and that is what is often in highest demand in management, but it does little to advance our understanding of basic processes. On the other hand, theoretical models, although they may not rapidly provide predictive ability,

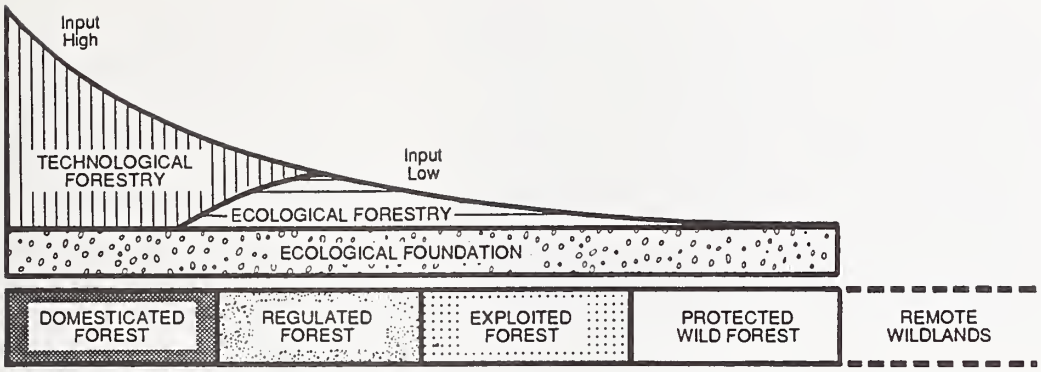

Figure 1—Schematic spectrum of the levels of forest management and degree of cultural control of the forest. (Adapted from Stone 1975.)

rapidly advance our understanding. When empirical models fail to meet our expectations we adjust the coefficients in order to improve the fit. When theoretical models fall short we must question our basic assumptions, our theory.

Although the theoretical basis of habitat types is little more advanced than our own, the development of theory-based ecological models has done much to advance our basic understanding of ecological processes. Clearly there is a place for the future development of soil-vegetation, soil-site theory, and the development of models to test that theory. We likely cannot attain a complete understanding of these complex process level interactions in the absence of such theory.

WHERE FROM HERE?

In general our efforts to develop a good understanding of how soils influence vegetation distribution and growth in the Mountain West have been fragmented and feeble. However, they may have been appropriate in light of the intensity of forest management. Where do we go from here? Is there a demand for better soils information? Do we have the tools necessary to meet that demand in a timely manner?

My guess is that more site-specific management planning will be required on both public and private forest land in the Mountain West. I know that good soils information can be very valuable in such efforts. Soils information has several advantages over habitat type, land type, or similar information in such planning applications.

In the first place it is objective rather than subjective. It is subject to less personal bias on the part of the classifier/mapper, and it is independent of current use standards and product strictures. It can be reinterpreted endlessly using new suitability and feasibility constraints.

Second, it is best collected at a large scale, but it can be aggregated easily for use at smaller scales. This is important when some decisions will be regional in nature, while others will be stand or location specific. It is also essential for error-free incorporation into geographic information systems.

Third, it can be interpreted for a wide variety of uses. When soil units have been adequately defined and interpretations properly and thoroughly developed, soils information is equally valuable to the silviculturist, recreation manager, hydrologist, range conservationist, wildlife manager, fisheries biologist, and even archeologist. This universality is unmatched by any other kind of landscape information, save legal property descriptions.

This all means to me that there should and will be an increasing demand for good soils information. The question then is can we supply it in a timely manner? My suspicion is that the answer is no. We have neither the cadre of trained personal to collect it nor the basic understanding of what it is we should seek to supply.

What are the most important features of the soils of the Mountain West? Organic matter in the A horizon, texture of the B horizon, percent coarse fragments and depth to bed rock, drainage class, acidity, mineralizable nitrogen? How should we develop soil taxa, and how should these taxa be delineated on the landscape (mapped)? Should we emulate the Soil Conservation Service, Weyerhaeuser, International Paper, or the procedures used by Forest Service elsewhere in the Country? And who and how will we train the army of people necessary to collect, interpret, and catalog this mountain of information?

I know I don't have the answers to these questions, and I doubt that anyone here has them. Hopefully, one of the things that will come out of this meeting is a mechanism for addressing these questions. We have shown in the East, South, and Midwest that we can classify and map soils and develop soil interpretations that prove to be extremely valuable in natural resource management. Surely we can do the same in the West, but it won't be easy.

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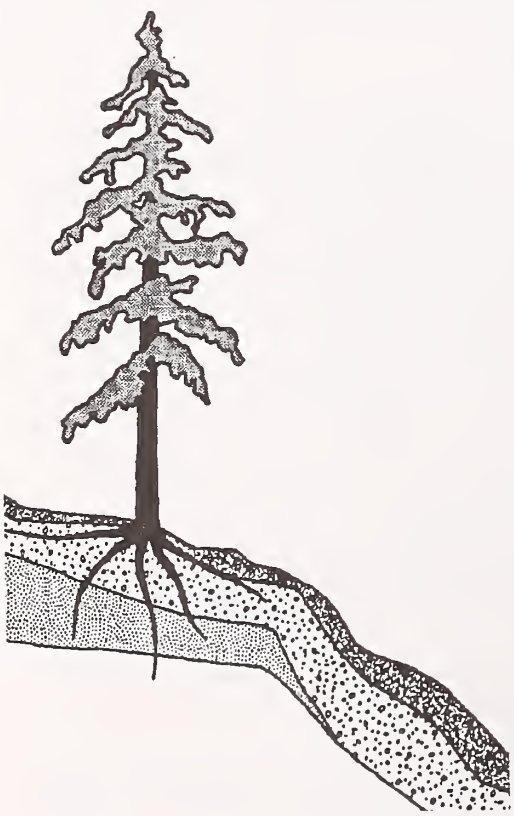
Soil Properties and Soil-Vegetation Interactions



DOMINANT SOIL FORMATION PROCESSES AND PROPERTIES IN WESTERN-MONTANE FOREST TYPES AND LANDSCAPES—SOME IMPLICATIONS FOR PRODUCTIVITY AND MANAGEMENT

Robert T. Meurisse
Wayne A. Robbie
Jerry Niehoff
Gary Ford

ABSTRACT

The principal soil orders in western-montane forests are Inceptisols, Alfisols, Andisols, and Mollisols. Soil moisture and temperature regimes strongly influence forest type distribution and productivity. The most productive and resilient forests are on soils with udic moisture and frigid temperature regimes. Soils with low water-holding capacity in ustic, xeric, and aridic moisture regimes and those with cryic temperature regimes are least productive and resilient. Soil organic carbon and nitrogen contents range from about 20,000 to more than 100,000 and 1,200 to 9,000 pounds per acre.

INTRODUCTION

The western-montane forests occupy a vast area of the western United States in which active mountain building, erosion processes, and weathering regimes interact to produce landscapes and ecosystems of great diversity, grandeur, and degrees of resilience to land disturbance. Geologically, land surfaces vary from very recent (several thousands of years) to very old (millions of years). Soil-forming processes operating on these diverse landscapes have formed heterogeneous soils with highly variant properties. The resulting soil-forest ecosystems are likewise heterogeneous in character and function and require site-specific understanding for making management interpretations that provide for ecosystem sustainability and health. Nevertheless, some important principles and concepts can be established to serve as guides for forest resource management.

Soil is the primary medium for regulating movement and storage of energy and water and for regulating cycles and availability of plant nutrients. Soil also provides anchorage, aeration, heat for roots, and is home for many decomposers and element-transforming organisms. Informed inquiry and understanding are critical for making sound decisions about effective and efficient use and management of these vital resources. The objectives of this paper are to: (1) characterize the dominant soil-formation processes and properties in the principal western-montane forest types and landscapes; (2) illustrate the major soil moisture and temperature regime gradients of these forest types; and (3) discuss some implications for ecosystem function, productivity, and management.

WESTERN-MONTANE FOREST TYPES AND EXTENT

The focus of this paper is on the western-montane forest types as described by Burns (1983) and the soils that support them. The principal forest types, their extent, and symbolic codes are in table 1. For further information about each of these types, refer to Agriculture Handbook 445. Additional information about the forest types particular to the Inland Northwest and their productivity, processes, and properties is provided by Harvey and others (1989).

PRINCIPAL KINDS OF SOILS

The most direct way to characterize the kinds of soil occurring in western-montane forests is to discuss the occurrence of soil orders. Soil orders are the highest taxonomic class in the United States system of soil classification (Soil Survey Staff 1975). They are differentiated by the presence or absence of diagnostic horizons or features that reflect differences in the dominant soil-forming processes that occurred. The recent amendment to the National Soil Taxonomy Handbook established a new soil order—Andisols (Soil Survey Staff 1989), making 11 soil orders. Hausenbuiller (1978) provides a more

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Table 1—Western-montane forest types and extent and their dominant soil moisture and temperature regimes

Code	Name	
		Million acres
	USTIC-UDIC Moisture Regime MESIC-FRIGID-CRYIC Temperature Regime	
SW PIPO	Southwestern ponderosa pine	11
PIPO/PSME v.gl	Ponderosa pine-Rocky Mountain Douglas-fir	20
MC/PIMO/THPL	Mixed conifer-western white pine-western redcedar	5
PICO	Lodgepole pine	13
PIEN/ABLA	Engelmann spruce-sub-alpine fir	10
	XERIC-UDIC Moisture Regime FRIGID-CRYIC Temperature Regime	
NW PIPO	Northwest Ponderosa pine	7
ABGR/PSME v.gl	Grand fir-Douglas-fir (Rocky Mountain)	6
LAOC	Western larch	3
ABMA/ABCO	Red fir-white fir	4
	USTIC-ARIDIC Moisture Regime MESIC-FRIGID Temperature Regime	
P-J	Pinjon-juniper	48

complete description of soil orders and the other taxonomic levels. Within the vast western-montane region, nine of the 11 soil orders occur, though several are rare. Only the highly weathered and leached Ultisols and Oxisols apparently are absent. Data are unavailable for precise estimates of the extent of soil orders. However, a review of selected soil survey reports and interpretation of data from "Soils of the Western United States" (Washington State University 1964) permits a reasonable estimate of relative occurrence (table 2). Probably the major difference between current estimates and the 1964 report is the relatively low occurrence of Spodosols. Many of the soils that were formerly thought to be Spodosols are now classified as Andisols, Alfisols, and Inceptisols.

Inceptisols are a diverse group of soils of intermediate development in humid and subhumid regions. They are mostly friable soils with moderate-to-high levels of organic carbon. Though many have high rock fragment content (more than 35 percent by volume) and have cryic temperature regimes, they usually are moderately deep and deep and are relatively fertile. Many are on unstable and active fluvial slopes in the Idaho Batholith.

Alfisols occur on a variety of landscapes, but most are stable or meta-stable surfaces. They support a wide range of forest types. Alfisols have relatively light-colored surface horizons and the dominant feature is the presence of an enriched subsurface clay layer. Most clays are the expanding 2:1 layer types. Ion exchange capacities are moderate to high and the soils have relatively high fertility. These soils dominate the Colorado Plateau in the southern part of the western-montane forests (Meurisse and others 1975).

Andisols are forming mostly in volcanic ejecta such as ash, pumice, cinders, and lava. These soils have a characteristic layer of volcanic ash or pumice, 14 inches (35 cm) to several feet thick over buried soils. These soils, and intergrades to other orders, are among the most productive of western-montane forest soils (Meurisse 1985, 1987). Andisols occur mostly in the northwestern and

Table 2—Estimated relative occurrence of soil orders in the western-montane forest region. The principal associated forest types are shown in their relative occurrence in each soil order

Soil order	Relative occurrence	Principal associated forest types
Inceptisols	3.0	PICO, PIEN/ABLA, PIPO/PSME, LAOC, ABMA/ABCO, ABGR/PSME
Alfisols	2.0	SW PIPO, PICO, PIPO/PSME, MC/PIMO/THPL, PIEN/ABLA, P-J, LAOC, ABMA/ABCO
Andisols	1.6	ABGR/PSME, MC/PIMO/THPL, NW PIPO, PICO, PIEN/ABLA, LAOC, SW PIPO
Mollisols	1.3	PIPO/PSME, SW PIPO, P-J, NW PIPO
Aridisols	.8	P-J
Entisols	.6	PICO, PIEN/ABLA, P-J
Spodosols	.5	PIEN/ABLA, PICO
Others	.2	—
	10.0	

northern parts of the western-montane region, on plateaus and gentle to moderately steep mountain slopes. Some are on steep and very steep slopes in northern Idaho and have a high erosion hazard.

Mollisols are characterized by thick, dark-colored surface horizons with at least 1 percent organic matter in the surface mineral soil layers. They occur mostly in the relatively dry western-montane forests and often have an abundance of grass and forbs in the understory. Many of the forests transitional to pinyon-juniper forests and grasslands occur on these soils. They occur mostly on stable mountain slopes and high plateaus.

The Aridisols occupy the driest parts of the region, and the principal forest is the pinjon-juniper type. Entisols are forming in parent materials of recent origin, in cold climates, and on some unstable land surfaces where erosion prevents development of diagnostic features. Spodosols are strongly acid and have low levels of base cations. They have predominantly cryic temperature regimes and support Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), and lodgepole pine (*Pinus contorta*) forest types.

SOIL MOISTURE AND TEMPERATURE REGIME GRADIENTS OF FOREST TYPES

Available soil moisture and soil temperature regimes are two overriding variables affecting the distribution, species mix, and productivity of western-montane forests. Soil moisture and temperature regimes are internal soil properties and are described in the Soil Taxonomy Handbook (Soil Survey Staff 1975) and by Hausenbuiller (1978). Generalized descriptions of these regimes are in table 3. The distribution of the principal western-montane forest types, by soil moisture and temperature regimes, is illustrated in figures 1 and 2. The size of the delineation is proportional to the distribution of the forest type according to these regimes and mean annual precipitation. It does not necessarily relate to aerial extent. For example, the ponderosa pine (*Pinus ponderosa*)/Rocky Mountain Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) type is the most extensive, except for the pinyon-juniper type (table 1). Yet, its soil moisture and temperature regime and mean annual precipitation range is relatively narrow.

Table 3—Soil moisture and temperature[1] regimes in western-montane forests

Soil moisture regimes		Soil temperature regimes	
Name	**Description**	**Name**	**Description**
Xeric	The soil moisture control section is dry for 45 or more consecutive days within the 4 months after the summer solstice. Typified in Mediterranean climates where winters are moist and cool and summers are warm and dry.	Mesic	The mean annual soil temperature is 8 °C (47 °F) or higher, but lower than 15 °C (59 °F). The difference between mean summer and winter temperature is more than 5 °C (9 °F).
Ustic	Soil moisture control section is dry for 90 or more cumulative days, but is moist in some part for more than 180 cumulative days. Soil moisture is limited, but available when conditions are suited for plant growth.	Frigid	The mean annual soil temperature is lower than 8 °C (47 °F) and the difference between the mean winter and summer soil temperature is more than 5 °C (9 °F).
Udic	Soil moisture control section is not dry for as long as 90 days (cumulative) and not dry for as long as 45 consecutive days in the 4 months following the summer solstice. Stored soil moisture plus rainfall ≥ ET amount.	Cryic	Soils have a mean annual temperature higher than 0 °C (32 °F) but lower than 8 °C (47 °F) and a mean summer temperature lower than 8 °C (47 °F) with an 0 Horizon.
Aridic	Soil moisture control section is dry more than half the time that the soil temperature is above 41 °F. Never moist for 90 consecutive days when soil temperature is above 47 °F.		

[1]Measured at 50-cm [20 inches] depth.

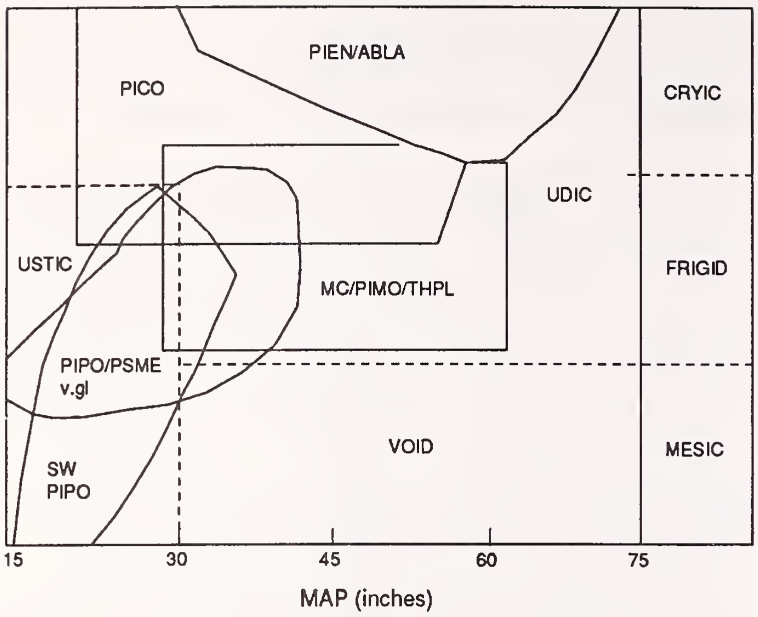

Figure 1—Distribution of western-montane forest types in ustic and udic soil moisture regimes as a function of mean annual precipitation, and mesic, frigid, and cryic soil temperature regimes.

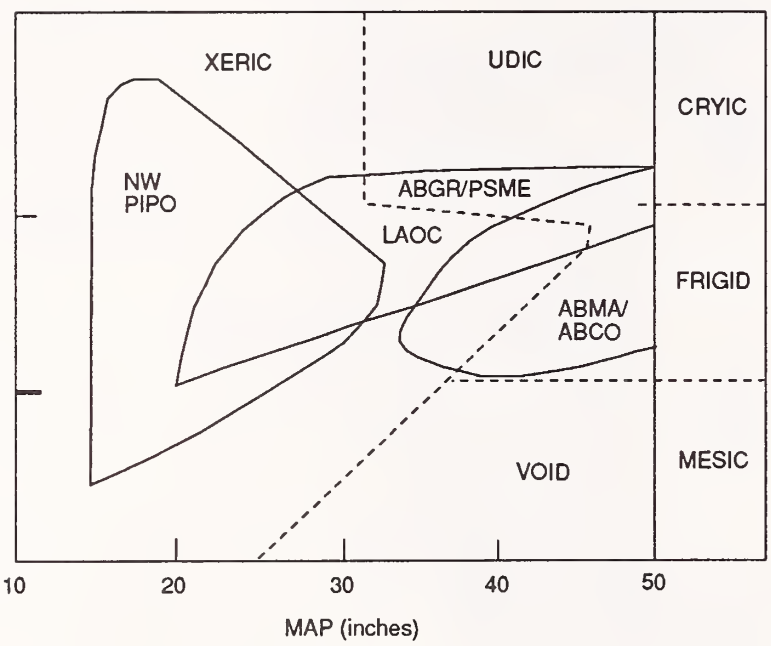

Figure 2—Distribution of western-montane forest types in xeric and udic soil moisture regimes, as a function of mean annual precipitation, and mesic, frigid, and cryic soil temperature regimes.

It is mostly on ustic and frigid regimes, but some are mesic and cryic and some are udic. The mixed conifer/western white pine (*Pinus monticola*)/western redcedar (*Thuja plicata*) type is on neither extremely cold nor very dry soils. It is generally the most productive of all the types in the region and is mostly on Andisols and Alfisols. At culmination of mean annual increment (CMAI), growth rates usually range from about 85 percent to about 150 ft^3/acre/yr (Burns 1983; Donaldson and others 1982; Donaldson and others in press).

The grand fir (*Abies grandis*)/Douglas-fir and western larch (*Larix occidentalis*) types occur in xeric and udic moisture regimes and usually are on soils derived from volcanic ash (Andisols). Many of these soils, such as the Tolo series, currently are classified as having xeric moisture regimes. However, data from Klock (1980) and Geist and Strickler (1978) suggest that many of these soils probably have an udic moisture regime. When these soils are at elevations higher than 4,300 feet and mean annual precipitation is more than 40 inches, moisture regimes may be udic. Where there is little volcanic ash, less than 7 inches (18 cm), these forest types are largely on soils with xeric moisture regimes.

Temperature regimes are mostly frigid, but some are cryic. The Andisol and Andic intergrade soils in these forest types have high readily available water holding capacities (Geist and Strickler 1978). This is a major factor contributing to relatively high productivity of 65 to 125, or more, ft^3/acre/yr at CMAI (Meurisse 1985, 1987). The northwest ponderosa pine forest (NW PIPO) is clearly the dominant xeric regime type. Though not shown in figure 2, some of the lodgepole pine type in association with the NW PIPO type also is xeric. Much of the NW PIPO type is on soils with cryic temperature regimes. The cryic soils under the NW PIPO type are mostly the pumiceous Andisols of the central Oregon plateau (Carlson 1979; Meurisse 1985, 1987). These soils are known to exhibit unique thermal properties which affect their management (Cochran 1975). This type is also on some soils with mesic temperature regimes. Plant available water-holding capacity is an important factor governing productivity on soils with a xeric moisture regime.

The lodgepole pine and Engelmann spruce/subalpine fir types are dominantly on soils with cryic temperature regimes and with udic moisture regimes. Thus, soil temperatures are a major limiting variable for tree growth. The southwest ponderosa pine forest type is mostly on soils with frigid and mesic temperature regimes and ustic moisture regimes. Soil water often is limiting plant growth and plant available water-holding capacity of the soils is critical. Some of the southwestern ponderosa pine forest has udic moisture regimes and is the most productive in this type (Meurisse and others 1975).

SOIL FORMATION PROCESSES

Soils form as the product of five major factors including climate, parent material, topography, organisms, and time (Jenny 1941). In western-montane forests, each of these factors assumes paramount importance in some part of the region. For example, continental and alpine glaciation have left mostly coarse-grained till deposits in the northern part of the region and at high elevations in the Rocky Mountains. Many soils from these parent materials have very dense layers, which are restrictive to both roots and water. The eruption of Mount Mazama, about 6,700 years before present, deposited volcanic ash, cinders, and pumice on much of the northern and western part of the region from the Cascades in Oregon and Washington to British Columbia, western Montana, and central to northern Idaho. Soils forming in this material, though very young, have very low bulk densities (0.5 to 1.0 g/cc) and have accrued relatively high amounts of organic matter (fig. 3). Thus, they are very favorable

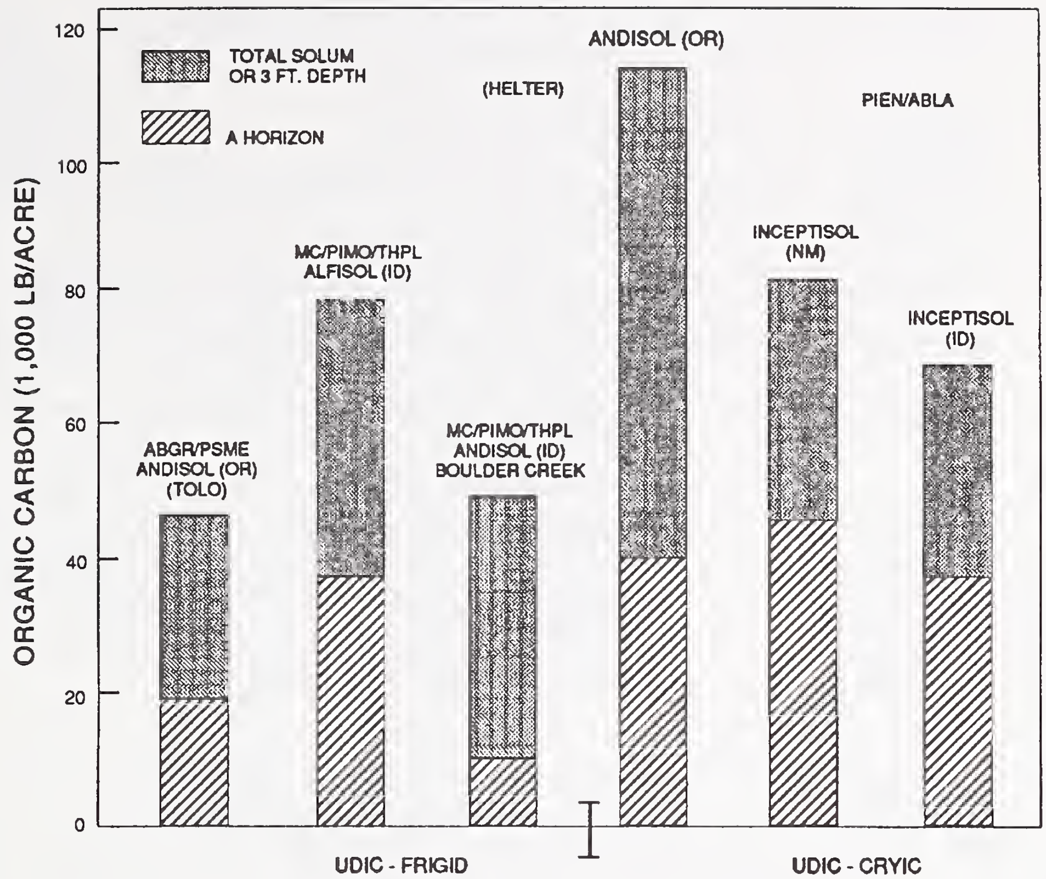

Figure 3—Mineral soil organic carbon content of representative soils and forest types in udic moisture and frigid and cryic temperature regimes. Source: Tolo and Helter adapted from Geist and Strickler (1978); others from unpublished National Cooperative Soil Survey Laboratory data.

for root development and water movement and storage. The southern part of the region usually has very old surfaces of sedimentary origin. The resulting Alfisols are among the most weathered soils in the western-montane forests.

The associated weathering processes of oxidation and reduction, hydration, mass flow, and ion exchange reactions result in soils with particular characteristics. However, soils are open-ended systems with many subprocesses and reactions. These processes of ongoing soil formation include: (1) additions to the soil; (2) losses from the soil; (3) transformations of material within the soil; and (4) translocations within the soil (Buol and others 1972). Table 4 illustrates many of these soil-forming processes which are active in western-montane forest soils. While many combinations, or even all, of these processes can occur to some degree in any soil, one or a few are dominant under specific conditions. Each soil order is the result of dominance by certain processes. For example, podzolization is dominant in Spodosols, but littering, illuviation, and eluviation also are important.

Losses from soils are from either erosion or leaching. In western-montane forests, leaching losses generally are minimal. Even in the temporary absence of forest cover, leaching losses seldom are important because of soils with relatively high ion exchange capacities and relatively dry soil conditions especially in the aridic, xeric, and ustic moisture regimes. Leaching losses may be important on some of the more moist sites with cryic temperature regimes, especially when vegetation is removed. Erosion losses are variable and can be accelerated with improper practices. This is particularly true on slopes greater than 40 percent and on soils with relatively low surface organic matter contents or moderate-to-low infiltration rates.

Additions to soils occur through littering, melanization, and cumulization. Wind-blown materials from adjacent arid regions represent a cumulization process that recycles bases such as calcium, magnesium, and potassium onto nearby forest soils. This gives rise to soils with moderate-to-high base status such as some Alfisols, Inceptisols, and Mollisols. In the absence of this process, some of these soils may advance in weathering stages to a degree that these nutrients would become limiting.

Littering, melanization, and pedoturbation are important processes in many soils, but especially in forested soils. Accumulation and mixing of organic matter are important for providing nutrients, ion exchange, aggregate stability, and water-holding capacity. Accumulation of organic matter also provides surface protection against raindrop splash, reducing the potential for erosion. Organic matter also serves as an energy source for organisms that perform many important functions in soils (Jurgensen and others 1979; Richards 1987). But these processes are only a part of the interaction of several processes that affect accumulation and cycling of carbon and nitrogen in soils. This includes the transformation

Table 4—Soil forming processes common in western-montane forest types

Translocation Within a Soil Body	
Eluviation	Movement out of a soil layer
Illuviation	Movement into a soil layer
Decalcification	Removal of calcium carbonate from a horizon
Calcification	Accumulation of calcium carbonate in a horizon
Lessivage	Mechanical migration of mineral particles from A to B horizons
Pedoturbation	Biological, physical (tree throw, freeze/thaw-wet/dry cycles), cycling of soil materials (homogenizing)
Podzolization	Chemical migration of aluminum and iron or organic matter—concentration of silica in the layer eluviated
Laterization	Chemical migration of silica out of the soil with concentration of sesquioxides
Transformation of Material Within a Soil Body	
Decomposition	Breakdown of mineral and organic materials
Synthesis	Formation of new particles of mineral and organic species
Humification	Transformation of raw organic material into humus
Mineralization	Release of oxide solids through decomposition of organic material
Braunification	Progressive oxidation and hydration giving the soil
Ferrugination	mass brownish, redish brown, and red colors
Additions to a Soil Body	
Cumulization	Aeolian/hydrologic additions to soil surface
Melanization	Darkening by admixture of organic matter
Littering	Accumulation of O.M. on the mineral surface
Losses from a Soil Body	
Leaching	Washing out of soluble material from the solum
Erosion, surficial	Removal of material from the soil

process of mineralization, which is critical to making nitrogen and sulfur, and to a lesser extent phosphorus, available for plant growth (fig. 4). A thorough discussion of these processes in soil ecosystems, including the role of soil biota, is presented by Richards (1987). The principal source of soil organic carbon is from plant litter (detritus), roots, and decomposer organisms. Over half the soil organic matter comes from roots, mycorrhizae, and other fungi (Cromack and others 1979; Sollins and others 1980).

The rates and magnitudes of organic carbon and nitrogen cycling and accumulation vary widely among the many types of soils in western-montane forests. Some examples of carbon and nitrogen accumulation for representative soils are in figures 3 and 5 through 7. These processes are subject to both natural (fire, wind, volcanic eruptions) and anthropogenic influences. Such influences can have temporal changes that either increase or decrease the process rates and magnitudes. Thus, it is important to gain an understanding of these processes and how they function in given soil ecosystems in order to manage for sustainability of desired forest types.

In the most arid parts of the region, calcification is an important process. It occurs where soil moisture is insufficient to move calcium through the soil profile. Calcium accumulations are root restrictive, and tree roots seldom penetrate layers of calcium carbonate. While juniper appears to have low sensitivity, ponderosa pine is more sensitive. Thus, regeneration and tree growth are hampered.

SOIL PROPERTIES IMPORTANT FOR PRODUCTIVITY AND MANAGEMENT

To understand the function of soil properties in relation to productivity and management practices, it is necessary to understand concepts of productivity.

General Functional Model of Productivity

The Soil Science Society of America (1978) defines soil productivity as: "The capacity of a soil in its normal environment, for producing a specified plant, or sequence of plants, under a specified system of management."

Productivity is governed by the magnitude and rates of change of several state variables. The magnitudes and dynamics vary in space and time, are interactive among each other, and are affected by management practices both locally and globally. A general functional model of the state (fundamental) variables governing productivity is: P = F (SM.SN.SA.L.H.V), where P is productivity as a measure of carbon fixation. Net primary productivity, though difficult to measure, is a complete measure of ecosystem dry-matter production. It is a useful measure for comparing different ecosystems (Grier and others 1989).

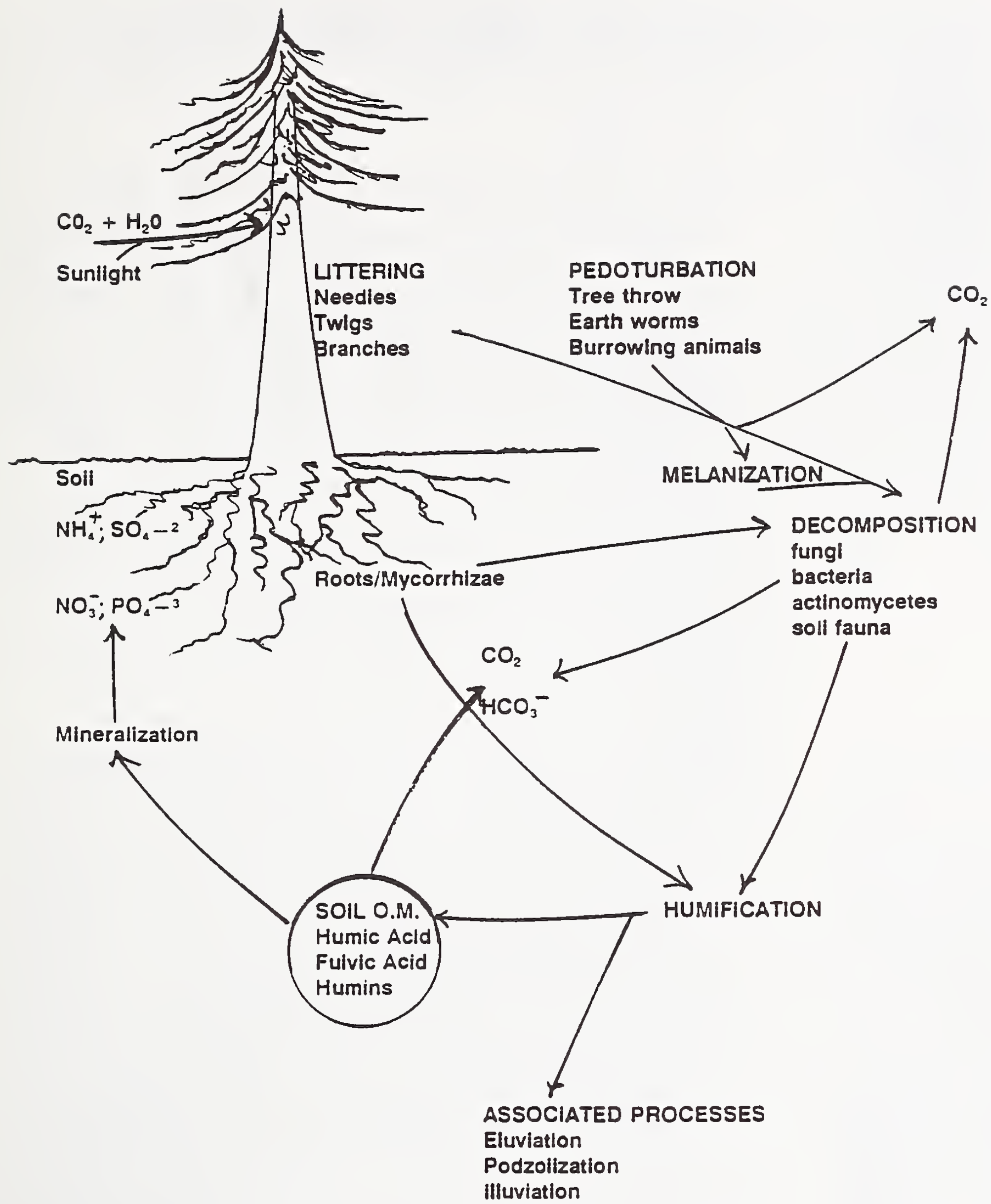

Figure 4—Dominant soil formation processes involving carbon cycling and accumulation in forest soils. The mineralization process releases nitrogen, sulfur, and phosphorus from soil organic matter in forms available for plant use.

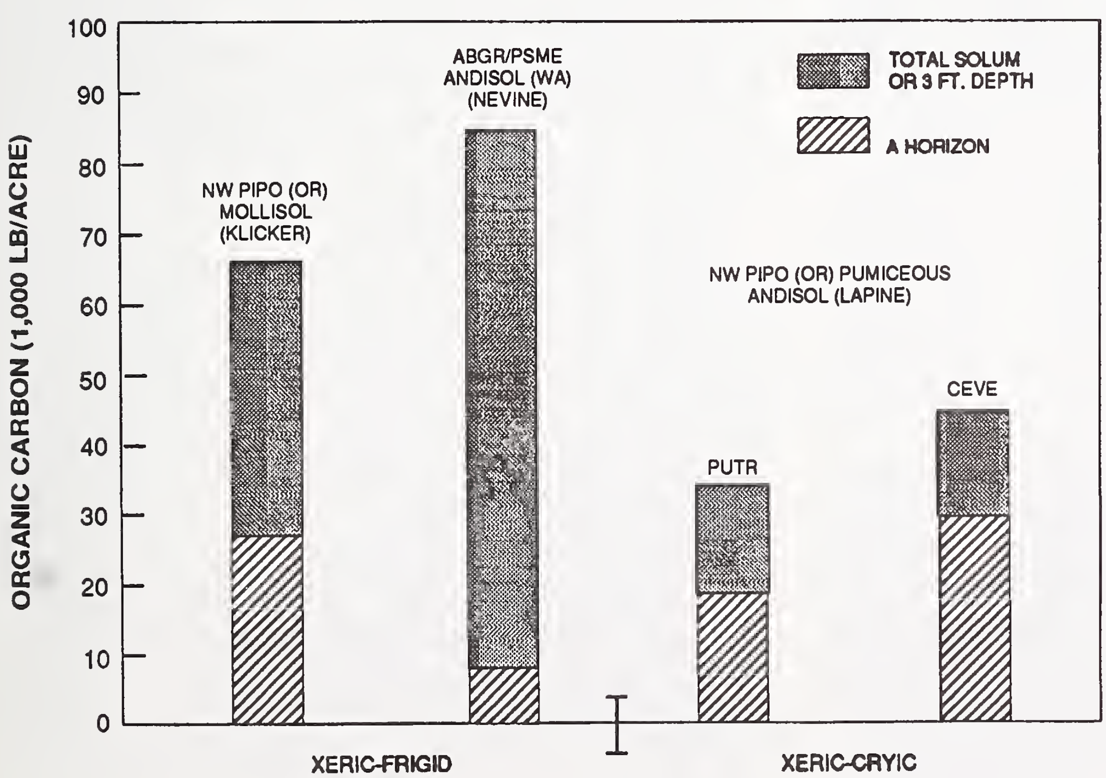

Figure 5—Mineral soil organic carbon content of representative soils and forest types in xeric moisture and frigid and cryic temperature regimes. Source: Klicker adapted from Geist and Strickler (1960); Nevine from unpublished National Cooperative Soil Survey Laboratory data; LaPine from Dyrness (1960).

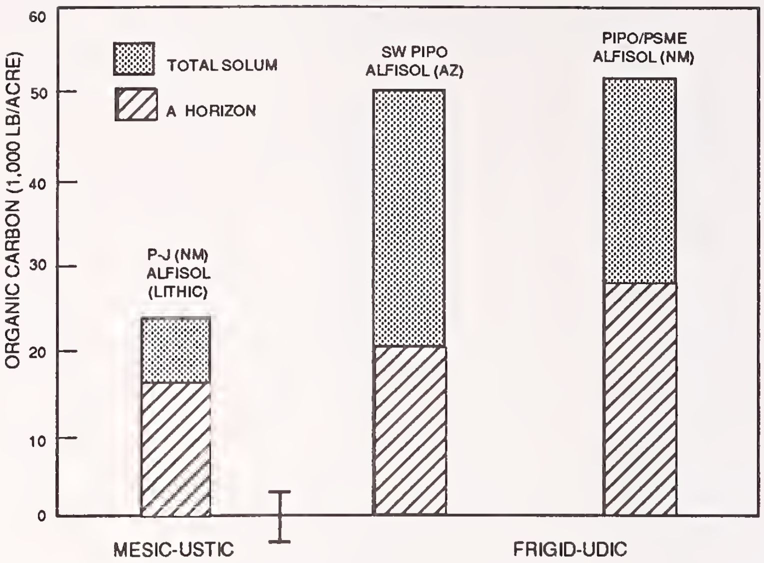

Figure 6—Mineral soil organic carbon content of representative soils and P-J SW PIPO and PIPO/PSME, forest types in ustic and udic moisture and mesic and frigid temperature regimes. Adapted from unpublished National Cooperative Soil Survey Laboratory data.

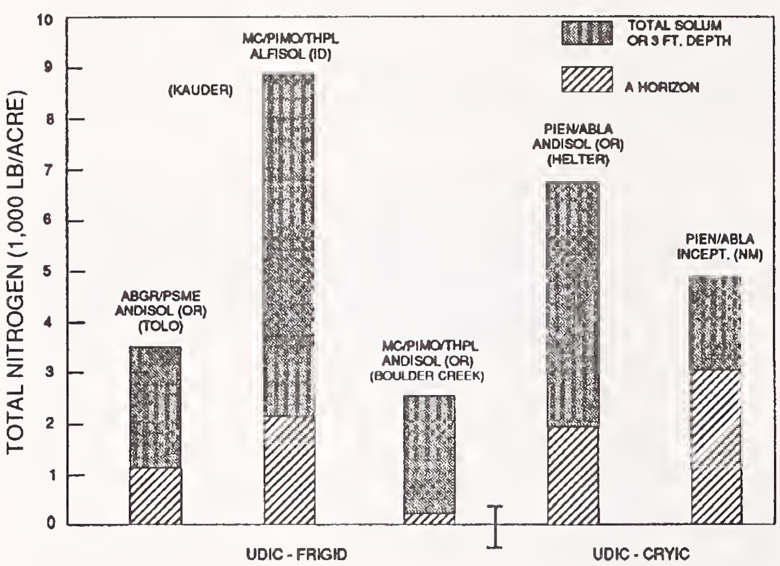

Figure 7—Mineral soil total nitrogen content of representative soils and forest types in udic moisture and frigid and cryic temperature regimes. Source: Tolo and Helter adapted from Geist and Strickler (1978); others from unpublished National Cooperative Soil Survey Laboratory data.

Other measures of practical use include site index and periodic increment per unit area. The state variables are: SM (available soil moisture), SN (available soil nutrients), SA (soil aeration), L (light quality and quantity), H (heat), and V (vegetation—species and genetic traits). Site capacity factors (state variables) most easily affected by management activities are soil moisture, nutrients, and aeration. The inherent ecosystem properties of soil moisture and nutrient supply, aeration, and heat are a function of long-term soil processes and are principal productive function regulators. Many soil properties interact to affect each of these state variables. Productivity can increase, decrease, or remain unchanged depending on how the state variables are changed.

Effects of Some Key Soil Properties

Several soil properties are paramount in their influence on available soil moisture, available soil nutrients, and aeration. These properties also are important in regional characterization of soils and in hydrologic processes.

Available Soil Moisture—Soil moisture regimes characterize the long-term general moisture supply. Properties such as soil depth, coarse fragment content, organic matter content, and soil texture and porosity regulate water infiltration, movement, storage, and release of water to plants. For example, clay soils hold large quantities of water per unit volume, but they also hold water at very high energies so that availability to plants is restricted. At the other extreme, low-density, high-porosity volcanic ash and pumice soils hold high amounts of water that is readily available at low energies (Geist and Strickler 1979). Management practices that cause soil compaction and displacement reduce infiltration rates and soil volume. Thus, plant available water supply also is reduced. Shallow soils and skeletal soils (greater than 35 percent rock fragments) have lower available water supply than moderately deep and deep soils with few rock fragments.

In general, the soils with udic moisture regimes and Andisols have the highest potential to supply plant available moisture in western-montane forests. Plant available water-holding capacity of these Andisols often is about 15 to 22 inches (Dyrness 1960; Geist and Strickler 1979). Table 5 illustrates some soil physical properties, including available water-holding capacity, for representative soil series or subgroups and their associated forest types. These soils are representative of Alfisols, Inceptisols, Andisols, and Mollisols and reflect the wide range of bulk density and available water-holding capacity of western-montane forest soils. Soils with inherent bulk density of 1.6 g/cc and greater usually restrict root development and penetration. Therefore, access to soil moisture is limited at those sites. Soils with these conditions usually are forming in dense glacial till or are some Alfisols that have fragipans or very high clay contents.

Available Soil Nutrients—Nutrient supply to plants is a function of several soil properties, including volume of root occupancy, mycorrhizae-root relationships, nutrient reservoirs, ion exchange properties, nutrient balance, and acidity. Soil texture, clay mineralogy, and organic matter content strongly influence ion exchange capacities, nutrient reservoir, mycorrhizal populations, and nutrient balance. Most western-montane forest soils are moderately acid to mildly alkaline (pH range of 5.6 to 7.8) and have base saturation of at least 35 percent. There are some soils, such as the Vay series, with base saturation of less than 35 percent. Such soils may be very strongly acid (pH 4.5 to 5.5). These are mostly cryic soils where leaching is prominent. Under these conditions, nutrients can be limiting plant growth. Under moderately alkaline conditions (pH 7.9 to 8.4), which may occur in soils with aridic and some ustic moisture regimes, nutrients also may be limiting.

Table 5—Some selected physical properties of representative soil series or soil subgroups by forest type. Bulk density of the subsoil is for the B horizon of highest density or for C horizons if there is no B horizon

Representative soil taxa	Forest type	Depth to restrictive layer	Bulk density Surface	Bulk density Subsoil	AWHC[1]
		Inches	----- *g/cc* -----		*Inches*
Eutric Glosoboralf(AZ)	PIPO/PSME	≥ 60	1.2	1.8	8 - 10
Mollic Eutroboralf(AZ)	SW PIPO	40 - 60	1.3	1.4	8 - 10
Lithic Haplustalf(AZ)	P-J	< 20	1.2	1.3	0.9 - 1
Dystric Cryochrept(NM)	PIEN/ABLA	40 - 60	1.4	1.3	2 - 3
Vay Series(ID)	PIEN/ABLA	40 - 60	.9	1.6	5 - 6
Boulder Creek Series(ID)	MC/PIMO/THPC	≥ 60	.8	.9	7 - 8
Kauder Series (ID)	MC/PIMO/THPC	35 - 40	.9	1.7	8 - 10
Tolo Series(OR)	ABGR/PSME	≥ 60	.7	.8	15 - 22
Nevine Series(WA)	ABGR/PSME	40 - 60	.6	.8	6 - 7
Nard Series(WA)	ABGR/PSME	≥ 60	1.6	1.7	12 - 13
LaPine Series(OR)	NW PIPO	≥ 60	.6	.5	15 - 22
Klicker Series(OR)	NW PIPO	20 - 40	.9	.9	3 - 4

[1]Plant available water-holding capacity.

Cation exchange capacities vary widely, but generally are adequate to retain an abundant supply of cations. Cation exchange capacities of Alfisols, Mollisols, and Aridisols usually are the highest of western-montane forest soils and range from 10 to more than 50 meq/100g. Some soils with low cation exchange capacities, such as some ashy and pumicious materials, have high internal porosity and are able to retain cations even after severe wildfire (Grier 1975). Soils with sandy loam and coarser textures, often from glacial till, usually have cation exchange capacities less than 10 meq/100g. When these soils occur in udic moisture regimes, nutrient losses can be significant, especially when vegetation is removed.

Nitrogen is the nutrient required in greatest quantity by trees and its abundance in the soil is a major factor in site quality. Nitrogen is known to be limiting in many western-montane forest soils (Cochran and Vander Ploeg 1988; Moore 1988). Total nitrogen, though very important, is only part of the story. More important, its availability when root activity is high is critical. Thus, the process of mineralization, which is influenced by soil moisture and temperature, and their effects on bacterial activity, determines whether nitrogen is available for plant uptake in amounts needed (fig. 4). This process also determines availability of sulfur and, to lesser extent, phosphorus. Thus, the soil organic matter content and nitrogen, sulfur, and phosphorus availability are inextricably linked. Figures 3 and 5 through 9 illustrate the contents of soil organic carbon and nitrogen in representative soils with different moisture and temperature regimes and forest types. The contents are for the mineral soil portion of the ecosystem. Organic carbon and nitrogen contents vary widely in western-montane forest soils. Also, the proportion in the surface layer varies measurably. These values are compared with the very high contents of about 250,000 to 725,000 lb/acre (280,000 to 812,000 kg/ha) of organic carbon and 15,000 to 38,000 lb/acre (16,800 to 42,560 kg/ha) of nitrogen in A and B horizons of some deep coastal Oregon Andisols under western hemlock (Meurisse 1972; 1976).

It is important to note that subtle differences may occur in soils under the same forest type, but with the presence of nitrogen-fixing understory plants. For example, figures 5 and 8 illustrate the Lapine series under ponderosa pine with an understory of bitterbrush (*Purshia tridentata*) and snowbrush (*Ceanothus velutinus*). Snowbrush fixes nitrogen in symbiosis with a bacterium in amounts of about 10 lb/acre/yr to 70 to 100 lb/acre/yr (Harvey and others 1989; Wollum and Davey 1975). Bitterbrush reportedly fixes less than 1 lb/acre/yr (Dalton and Zobel 1977; Haines and DeBell 1980). Nitrogen and organic carbon contents of surface soils are measurably increased in the presence of some nonleguminous N-fixing shrubs. Review of the data in figures 3 and 5 through 9 reveals some general patterns of organic carbon and nitrogen content. Generally, Andisols, especially those that are ashy and medial, have the highest contents of organic carbon and nitrogen in western-montane forest soils. The proportion in the surface to the total is highest in the weakly developed pumiceous Andisols. The soils in udic moisture regimes and frigid and cryic temperature regimes usually have the highest organic carbon and nitrogen content. The soils from more southerly latitudes generally have the least amount of organic carbon and nitrogen. Soils in ponderosa pine forests, except for Mollisols, usually are among the lowest in organic carbon and nitrogen content. Those soils with a relatively large proportion of carbon and nitrogen in the surface, relative to the total, generally are less resilient and more sensitive to soil disturbance, especially soil displacement.

Soil Aeration—The soil atmosphere or gaseous phase is one of the three phases in nature represented in soil. The other two are soil matrix or solid phase and soil solution or liquid phase. Of the three, soil air is least discussed. Yet its importance in soil ecosystem function should not be underestimated. Soil aeration is needed for gas exchange with respiring roots. Aeration also effects soil biota and heat-transfer processes. Fortunately, most forest soils have high natural porosities, especially macro

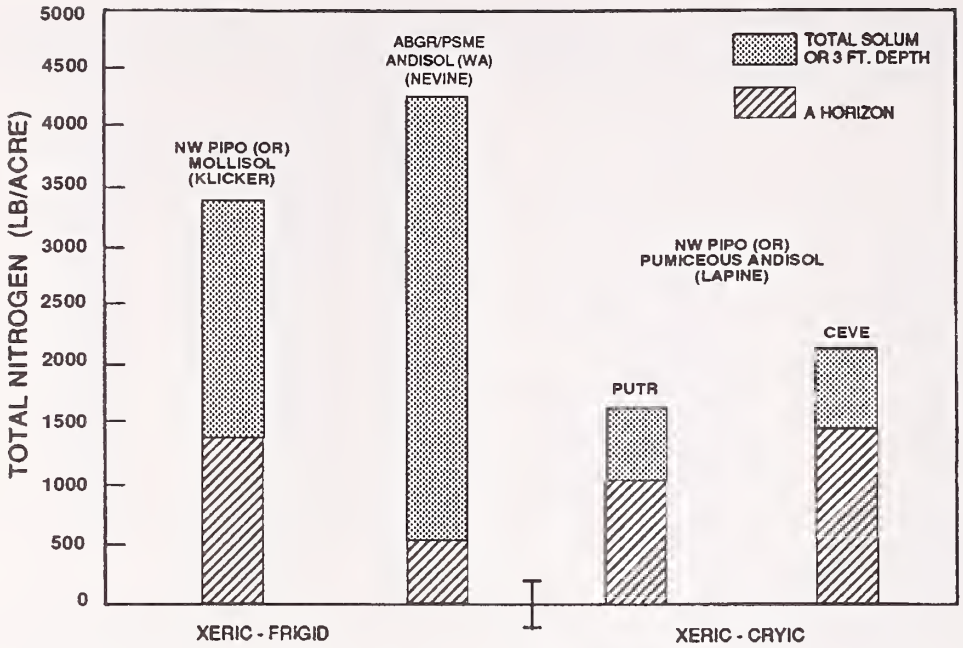

Figure 8—Mineral soil total nitrogen content of representative soils and forest types in xeric moisture and frigid and cryic temperature regimes. Source: Klicker adapted from Geist and Strickler (1978); Nevine from unpublished National Co-operative. Soil Survey Laboratory data; LaPine from Dyrness (1960) and unpublished National Cooperative Soil Survey data.

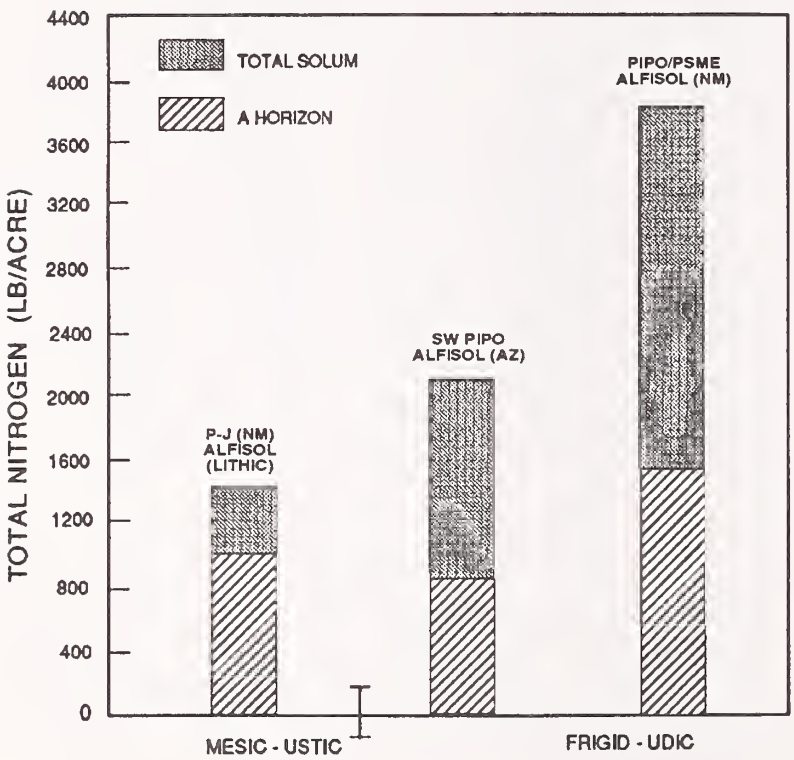

Figure 9—Mineral soil total nitrogen content of representative soils and P-J SW PIPO and PIPO/PSME forest types in ustic and udic moisture and mesic and frigid temperature regimes. Adapted from unpublished National Cooperative Soil Survey Laboratory data.

or air-filled pores and relatively low bulk densities (mass per unit volume). Indeed this is a characteristic of most temperate forests. However, soil porosity also is easily affected by forest management activities, especially where heavy ground equipment is used and concentrations of animals persist. Soil aeration is a function of soil porosity and drainage. The majority of western-montane forest soils are well drained or excessively drained and well aerated. Some soils with dense till layers or fragipans may have temporary or perched water tables with reduced aeration, but they are uncommon. Soil compaction and puddling from management practices is likely to have the greatest effect on aeration. The compaction process and effects in Pacific Northwest forests has been discussed by Froehlich and McNabb (1984). Similar effects can be expected in western-montane forests. Harvey and others (1989) discussed these processes and effects for mixed conifer and pine forests of the inland northwest where Andisols are common soils. Clayton and others (1987) found declines in one or more growth attributes in 15- to 25-year-old ponderosa pine and lodgepole pine stands on Inceptisols with a volcanic ash surface layer. The declines were associated with increased bulk density, increased penetration resistance, and lateral soil displacement.

Soil bulk density is one of the most common indirect measures of porosity and soil strength in the field. Air permeameters have been used to measure macro porosity (Steinbrenner 1959). Soils with low bulk density have correspondingly high porosity. Data in table 5 illustrate some of the variation in soil bulk density for surface and subsoil horizons. While most of the soils have relatively low surface bulk density, several have very high subsoil

density. Such soils, even though deep, have restricted root development. Soils with lowest densities are the Andisols and Andic intergrades. Some of the more dense and less porous soils are the Alfisols and other soils underlain by firm glacial till. The importance of low bulk density and high porosity on forest productivity and management of Andisols has been discussed by Meurisse (1985, 1987).

RESILIENCE OF SOIL ECOSYSTEMS

Soils are characterized by their composite set of properties and processes. The ability of soils to sustain long-term forest productivity is highly variable and is a function of their properties and processes. Similarly, soils vary in their resilience or ability to rebound or recover from management activities or natural disturbances. The diversity of soil types in western-montane forests suggests there is also diversity of resilience. The hazards of soil damage from erosion—surface and mass failure—compaction, puddling, and displacement often are governed by the shape and locations on the landscape, physical properties, and the kind and timing of activities. Indeed, some western-montane soils have very high hazards for erosion (Megahan and others 1978; Megahan 1981).

Nevertheless, many western-montane forest soils have properties that impart high resilience and low sensitivity (Meurisse 1985, 1987; Meurisse and others 1975). Some of the important properties affecting resilience are soil organic matter in both surface and subsoil layers, effective soil depth, inherent erodibility, plant available water-holding capacity, texture, structure, cation exchange capacity, shrink-swell potential, moisture regime, and temperature regime. To assess potential for soil damage, resilience, and productivity potential it is necessary to have high-quality soil surveys and to make site-specific soil investigations by qualified soil scientists. In doing so, the full potential of the many kinds of western-montane forest soils can be realized and their production capacity can be sustained for future generations. Also, for those soils with high sensitivity or low resilience, management prescriptions can be designed to overcome many of the inherent limitations and avoid detrimental soil damage, which would require costly rehabilitation or loss of productive potential of our precious soil resources.

CONCLUSIONS

Western-montane forest soils and landscapes are highly variable. The soils are forming from the interaction of many processes and materials. Each soil possesses a characteristic set of properties and processes, which must be understood to assure their management for long-term sustainability. Nine of the 11 soil orders are present. Inceptisols, Alfisols, and Andisols, respectively, are most abundant and occur in about 70 percent of the area. Soil-moisture regimes of western-montane forests are dominantly ustic, xeric, or aridic. Because these regimes are relatively droughty, available water-holding capacity of the soils is very important for forest productivity and reforestation. Several forest types are in udic moisture regimes where soil moisture generally is not the most limiting factor for plant growth. Rather, soil temperature in cryic regimes and soil fertility limit plant growth.

In the northern part of the region and at high elevations, continental and alpine glaciation have influenced soil formation. Where dense till layers occur, water movement and root growth are restricted. The eruption of Mount Mazama and subsequent deposits of volcanic ash, cinders, and pumice generally has enhanced the rooting medium. Protection of the surface layer of volcanic ash is critical for sustaining productivity of many of the soils in these forests.

Soil organic carbon and nitrogen contents vary widely among forest types and climatic regimes. Generally, the higher the organic carbon and nitrogen contents the higher potential productivity and soil resiliency.

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WESTERN-MONTANE PLANT COMMUNITIES AND FOREST ECOSYSTEM PERSPECTIVES

Robert Steele
Robert D. Pfister

ABSTRACT

The vegetation of "western-montane" forests is outlined in the context of different climatic regimes that vary from inland maritime to continental. Soil / vegetation correlations are most evident in the more severe environments, particularly with granitic versus limestone substrates. In moderate environments, soil / vegetation correlations can be derived but with greater difficulty. In most cases, these correlations are strongest on a local basis; regional correlations are difficult to achieve. Weak soil / vegetation correlations often result from the way soils and plant communities are described and from the effect of compensating factors within plant communities. The problem of compensating factors is illustrated in three soil / vegetation studies. An ecological perspective toward correlating soils and vegetation is emphasized. Combining ecosystem classification with landtype mapping is considered a useful result of the ecological perspective.

INTRODUCTION

Plant communities of western-montane forests vary geographically and largely as a function of climate, topographic influences, and substrate. Site history is also involved. Western-montane forests experience two distinct climatic influences, a Pacific maritime climate occurring mostly during winter and a continental climate during the summer. The relative proportions of these two climatic influences vary geographically: the northernmost areas are predominately maritime and the southeasternmost areas predominately continental. Throughout this broad area, most western-montane forests fall within the Engelmann spruce (*Picea engelmannii*), western hemlock (*Tsuga heterophylla*), Douglas-fir (*Pseudotsuga menziesii*), and ponderosa pine (*Pinus ponderosa*) floristic provinces of Daubenmire (1978) and the Columbia, Sierran, and Rocky Mountain forest provinces of Bailey (1980).

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INLAND MARITIME VEGETATION/ CLIMATE RELATIONSHIPS

The Pacific maritime influence is most evident in northeastern Washington, northern Idaho, and northwestern Montana. It occurs in diminished and varied forms across much of eastern Washington and Oregon, in west-central Idaho, and along the east slope of the Cascades and Sierra Nevada mountains (fig. 1) in eastern California. These high-mountain ranges intercept much of the maritime moisture and create a rainshadow to the east that precludes forest growth over a vast area.

The moist Pacific air masses are carried inland by the prevailing westerlies and have a dominating influence on vegetation over much of the northern western-montane area (Daubenmire 1969). Extended periods of cloud cover and fog with prolonged cyclonic storms characterize the weather pattern from fall through late spring. During summer, the westerlies shift northward and a weak continental climate creates drought with few clouds during the latter part of the growing season.

The resulting environment is relatively mild in these forests, supporting many plant species that occur largely west of the Cascade-Sierran axis. Western hemlock, western redcedar (*Thuja plicata*), mountain hemlock (*Tsuga mertensiana*), western white pine (*Pinus monticola*), and Pacific yew (*Taxus brevifolia*) comprise much of the forest found in the core area of maritime influence (fig. 1). Beyond the core area these species decline; this provides greater opportunity for other maritime-dependent species such as western larch (*Larix occidentalis*) and grand fir (*Abies grandis*). On the east slope of the Sierras, Jeffrey pine (*Pinus jeffreyi*) and white fir (*Abies concolor* var. *lowiana*) occupy similar conditions. Even ponderosa pine, which occupies the driest forest sites throughout the area, is the variety *ponderosa* that has Pacific maritime affinities (Steele 1988). In Oregon (Franklin and Dyrness 1973), Idaho, Montana (Arno 1979), and possibly elsewhere, this pine's southern and eastern geographic limits approximate the extent of Pacific maritime flora.

INLAND MARITIME VEGETATION/ SOIL RELATIONSHIPS

Widespread soil/vegetation correlations may be more difficult to obtain in this inland maritime environment, than in harsher environments that are more restrictive to plant distribution. Much of this area, especially in Idaho

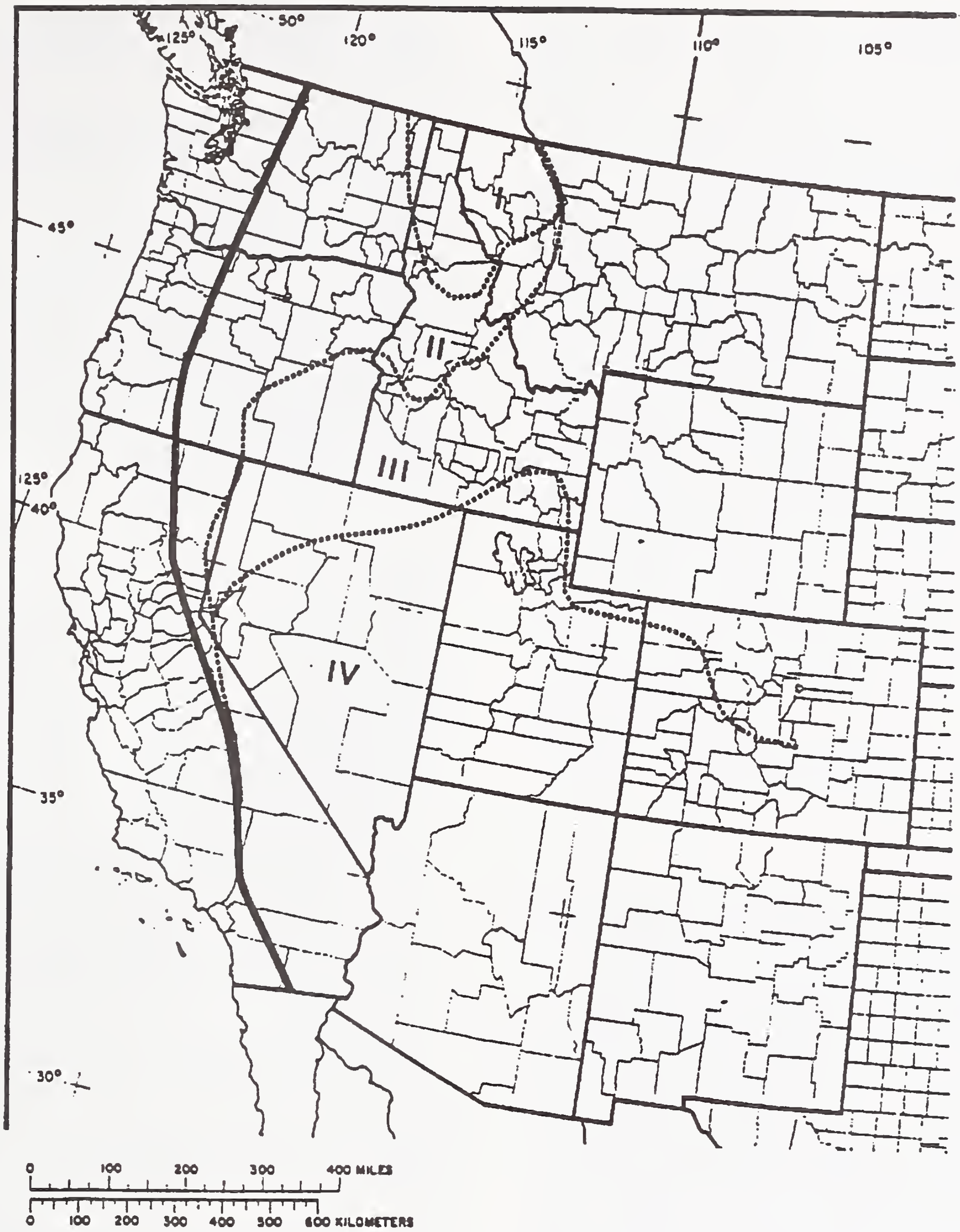


Figure 1—Approximate climatic regions in western-montane forests as suggested by distributions of indicative tree species. (I—core maritime, II—inland maritime, III—northern continental, IV—southern continental).

and northern Washington, also contains weakly developed forest soils such as those of granitic and aeolian (ashcap) origin as well as skeletal soils of several other rock types. The way soils and plant species have been classified can also lead to poor soil/vegetation relationships. A plant's ability to exploit compensating factors exacerbates these often poor correlations. Some major tree species such as grand fir (Daniels 1969) and Douglas-fir (Rehfeldt 1989) have a broad range of genetic diversity, and other plant species may also be genetically diverse, which may confound soil/vegetation relationships. Consequently, one should not always expect soil/vegetation correlations of inland maritime forests to be easily derived.

In central Idaho, for example, a Douglas-fir/ninebark (*Pseudotsuga menziesii/Physocarpus malvaceus*) association on basaltic soils is nearly identical to that on granitic soils. Grand fir associations are also similar across these two contrasting parent materials (Steele and others 1981). But toward the environmental extremes of the forest zone such as at lower timberline, soil-controlled differences in vegetation become more evident. For example, a ponderosa pine/bluebunch wheatgrass (*Pinus ponderosa/Agropyron spicatum*) association, particularly the forb component (Steele and others 1981), on basaltic soils is often different floristically from its counterpart on granitic soils.

On a local basis, some correlations have been derived. Neiman (1986) was able to identify differentiating soil characteristics in several highly similar grand fir and western redcedar associations. In the Garnet Mountains of west-central Montana, limestone and granitic substrates support notably different forest communities

(Goldin and Nimlos 1977). Other relationships may emerge as we explore soil/vegetation relationships in more detail.

CONTINENTAL VEGETATION/ CLIMATE RELATIONSHIPS

To the south and east of the inland maritime region lies an area having a stronger continental climatic influence, although the maritime influence remains evident during winter and spring in varying degrees (Bradley 1976). Here summer thunderstorms deliver moisture in downpours during July and August and, in some areas, into September. This moisture moves inland in the form of high-altitude air masses from the Gulf of Mexico and southern California coast resulting in convectional storms over the mountainous areas. Precipitation diminishes as these air masses move inland with only "dry lightning" often available to the inland maritime area of Idaho and western Montana. Rarely is there enough cloud cover to moderate the temperature for a significant period. The vegetation in general is quite different; most forest species of the inland maritime area are absent. Even the forest zone is more restricted, having been replaced at lower elevations by pygmy woodlands and chaparral.

Southern Area

In extreme southeastern Idaho, most of Nevada, Utah (except the Uinta Mountains), southern and western Colorado, Arizona, and New Mexico (fig. 1) there is a somewhat different mix of tree species than is found to the north. These forests are generally found above extensive woodlands of various pinyon, juniper, manzanita, and oak species. In some areas the interior variety of ponderosa pine (*Pinus ponderosa* var. *scopulorum*) forms extensive lower timberline forests, often with grass-dominated undergrowths (Hanks and others 1983). But in much of the area, adequate precipitation occurs only at elevations too cold for the pine seedlings to survive. Here Douglas-fir or quaking aspen (*Populus tremuloides*) usually forms lower timberline, although in some areas mixtures with limber pine (*Pinus flexilis*) or blue spruce (*Picea pungens*) may occur. The blue spruce also dominates certain topoedaphic situations such as alluvial or colluvial deposits, frost pockets, and riparian zones. With increasing elevation, and presumably more moisture, a few additional tree species become major forest components. Concolor fir (*Abies concolor* var. *concolor*) mixes with the Douglas-fir to form an extensive mid-elevation zone. In Arizona south of the Mogollon Rim, and in southwestern New Mexico, several pines (*Pinus strobiformis, P. leiophylla, P. arizonica, P. engelmannii*) and associated evergreen oaks (*Quercus hypoleucoides, Q. rugosa, Q. arizonica, Q. emoryi*) add diversity to these mid- and lower elevation forests (Bassett and others 1987; Layser and Schubert 1979). Many of these species represent a northern extension of Mexican flora.

At upper elevations, Engelmann spruce and subalpine fir (*Abies lasiocarpa*) form a distinct forest zone. Across its broad geographic range, this zone displays a diverse species composition in the undergrowth. Yet at the habitat type level (Moir and Ludwig 1979), some areas in Arizona and New Mexico are remarkably similar to the more continental areas of the northern Rockies. Also at upper elevations and often in topoedaphic situations the bristlecone pines occur. Minor forests of Rocky Mountain bristlecone pine (*Pinus aristata*) occur along the highest ridges from central Colorado to northern New Mexico. Engelmann spruce or Douglas-fir are often major associates of this bristlecone, but limber pine is notably sparse (DeVelice and others 1986). In Utah, Nevada, and extreme eastern California, Great Basin bristlecone (*Pinus longaeva*) occupies high, barren ridges and is often associated with limber pine and to a lesser extent Douglas-fir (Youngblood and Mauk 1985).

Northern Area

In northwestern Nevada, southeastern Oregon, most of southern and eastern Idaho, northern Colorado, the Uinta Mountains of Utah, and northward through Wyoming and Montana (fig. 1), the continental forest mosaic is somewhat different. This area lacks the extensive pinyon, juniper, and oak communities found to the south although stunted limber pine forms a pygmy woodland in central Montana (Pfister and others 1977) and northern extensions of Utah juniper (*Juniperis osteosperma*) appear in east-central Idaho and north-central Wyoming. The interior variety of ponderosa pine forms low-elevation forests (mainly east of the Continental Divide) and has diverse grass, sedge, and shrub-dominated undergrowths.

Like the southern area, lower timberline often occurs at elevations too high and cold for the pine. Then either Douglas-fir or quaking aspen form the lower timberline. Douglas-fir creates an extensive lower forest zone that includes limber pine and quaking aspen in some areas. Generally this zone merges directly with the Engelmann spruce-subalpine fir zone, since concolor fir is absent.

With increasing distance from the inland maritime area and with increasing elevation, spruce gains importance over subalpine fir. Engelmann spruce climax forests have been recognized in central Montana (Pfister and others 1977), east-central Idaho (Steele and others 1981), and Wyoming (Hoffman and Alexander 1976). In Montana, climax spruce occurs along the lower subalpine forest zone, reflecting hybridization with white spruce (*Picea glauca*) (Pfister and others 1977). Throughout much of these high-elevation forests, lodgepole pine (*Pinus contorta*) dominates the stand as a persistent but seral species. On a few severe sites lodgepole pine is thought to be climax (Hoffmann and Alexander 1976; Pfister and others 1977; Steele and others 1981). At upper timberline, whitebark pine (*Pinus albicaulis*) forms extensive open forest, but in some areas is limited by substrate. However, it is largely absent in north-central Wyoming and the Uinta Mountains of Utah where suitable granitic substrates exist.

CONTINENTAL VEGETATION/SOILS RELATIONSHIPS

Areas that lack a strong maritime influence present a more stressful environment, including winter dessication and extreme fluctuations in temperature, for many forest species. Soil/vegetation relationships appear more consistent for broader areas and some generalizations can be made. For instance in east-central Idaho, limber pine communities are often associated with volcanic and calcareous parent materials while the granitics and noncalcareous sedimentaries are occupied by whitebark pine and lodgepole pine communities. In the White Mountains of eastern California, many species, including Great Basin bristlecone pine, were shown to have strong affinity for a particular substrate (Marchand 1973). Likewise, in the Bighorn Mountains of north-central Wyoming (Despain 1973), much of the vegetation pattern is controlled by substrate. The most striking contrast in all of these studies is the effect of limestone versus granitic parent material on the vegetation.

Defining soil/vegetation relationships for the western-montane forests can be difficult. In general, they appear easier to differentiate in the more severe climatic regimes. In the less-severe inland maritime area they appear easier to achieve in the more severe habitats. Throughout the western-montane area, these relationships are most evident on a local rather than regional basis but are not consistent even on a local basis. There is still much to learn in the area of soils/vegetation relationships. Potential climax vegetation has been classified over much of the western-montane area (see Wellner 1989) and so have soils (Soil Survey Staff 1975). Developing ecologically sound linkages between the two is a logical next step.

SOIL/VEGETATION RELATIONSHIPS AT THE MICROSCALE LEVEL

The foregoing overview of vegetation and ecological relationships for western-montane forests leads us to a problem that continues to face both soil scientists and plant ecologists. That is factor interaction, or the principle of compensating factors. Three examples illustrate this problem:

1. In a study of soils and vegetation of the subalpine forests of Utah (Pfister 1972), complete vegetation and soil profile data were sampled on 24 sites. Major soils differences existed within and among the plant community types (table 1). The major controlling factors for vegetation distribution were obviously climate and microclimate, but differences in those factors were only weakly expressed in soil profiles.

2. An ongoing cooperative study between the Soil Conservation Service (SCS), U.S. Department of Agriculture, and the University of Montana is exploring the range of productivity within soil types, habitat types, and soil/habitat combinations. This data base consists of a large number of points covering the range of forest soils on private land and State forest land in Missoula County, MT. Soil profiles were described by SCS soil scientists and assigned to Soil Series. Habitat types and phases were identified using the Montana classification (Pfister and others 1977). Five site trees were sampled for the major species on each plot. A table comparing habitat types and soil types clearly demonstrated the Principle of Compensating Factors—correlation among the independent taxonomies was low.

The primary objective of the study is to test the hypothesis that the range in productivity within a habitat type or within a soil type could be reduced by subdividing one type by the other type. Mixed results are being obtained. For some combinations the cross classification sorts out high and low ends of the productivity range. However, for most types the range of productivity for the combination is about the same as the range for the individual soil type or habitat type. Therefore, we must be careful about generalizing, because the Principle of Compensating Factors is also operating on the variable of productivity. The site factors influencing the composition of a plant community or the taxonomic characteristics of a soil profile are not identical to those influencing productivity.

3. An example of soil-vegetation relationships was recently derived from the University of Montana's new geographic information system (GIS) that is operational for Lubrecht Experimental Forest. Two independent mapping efforts have produced separate layers that can be compared. Both mapping efforts were done at a scale of 1:15,840. The soil mapping was done by the SCS with

Table 1—Soil and habitat type relationships of plots in the subalpine forests of Utah (Ecological Response Units) (From Pfister 1972)

	Habitat type and phase					
	ABLA/BERE			ABLA/RIMO		
Soil subgroup	ABLA	BERE	RIMO	THFE	RIMO	ABLA/VASC
	Number of plots					
Dystric Cryochrept	—	1	—	—	2	3
Typic Cryochrept	1	1	—	1	—	—
Typic Cryumbrept	—	—	—	1	1	—
Entic Cryumbrept	—	1	—	—	1	—
Cryic Rendoll	1	—	—	2	—	—
Mollic Cryoboralf	1	—	—	1	—	—
Typic Cryoboralf	1	1	3	—	1	—

map units generally denoting slope phases of Soil Series (Nimlos 1986). Habitat types, to the phase level, were mapped by two graduate students as part of their wildlife research projects. From these two maps we can make comparisons at a detailed level of mapping where degrees of correlation should be most clearly expressed. Table 2 illustrates some of the key values from this exercise. The Principle of Compensating Factors was again clearly evident for the 2,804 forest acres. The habitat type level had 10 mapping units with 62 polygons; the soil series level had nine mapping units with 33 polygons; the overlay of these two maps produced 33 ecological land units and 109 polygons. The habitat type phase level had 15 mapping units with 103 polygons; the soil series slope phase level had 17 mapping units with 52 polygons; the overlay of these two maps produced 85 ecological land units with 199 polygons.

This exercise illustrates the scale and size of units that would be required to identify a homogeneous unit of land that would have similar climax vegetation potential, soil characteristics, and slope steepness, and therefore, similar responses to management. Is the average size of 14 acres too small for practical application? If we require larger units we must accept a greater amount of variation in vegetation potential, in soil characteristics, or both.

We have used these microscale examples to illustrate measured relationships from individual plots and detailed mapping. These kinds of examples are very useful to clarify concepts and terminology. Aggregating upward from detailed accurate maps and data bases is a very useful method to clarify discussions at broader levels of ecosystem classification and management.

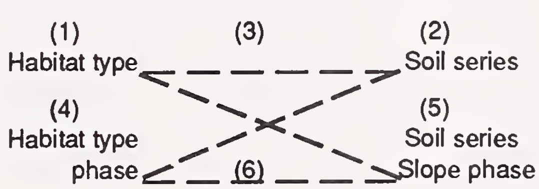
INTEGRATION FROM AN ECOLOGICAL PERSPECTIVE

This is a soils symposium, and the soils perspective is your common ground. Yet, you invited some plant ecologists with a different viewpoint to promote broadening of perspectives. One goal of a symposium like this should be to broaden each of our perspectives toward shared understanding and finding consensus—although in the process we will certainly not be able to avoid dealing with concepts that have polarized some individuals to the point where they would rather not talk to each other. Disagreement can be bad or good. It can lead to anger, frustration, lack of communication, and isolation. On the other hand, disagreement is essential as a stimulus for new ideas. The challenge is for us to handle disagreement as professionals.

We are probably all on common ground with a general understanding of ecosystem concepts. We all have a holistic perspective. We understand that ecosystems are complex, multifactorial, interacting and interconnected, as well as varying in time and space. Changes are determined

Table 2—Habitat types, soil types, and ecological land units (E.L.U.) for Bald Hills Unit, Lubrecht Experimental Forest[1]

	(1) Habitat type	(2) Soil series phase	(3) E.L.U. (H.T. x S.S.) phase	(4) Habitat type phase	(5) Soil series slope phase	(6) E.L.U. (H.T.P.x S.S.P.)
No. of types	10	9	33	15	17	85
No. of polygons	62	33	109	103	52	199
Average size polygons (acres)	60	85	26	11	14	14
Range in polygon sizes (acres)	3 to 362	3 to 433	3 to 380	3 to 66	3 to 174	3 to 173
Total forested acres	2,804	2,804	2,804	2,804	2,804	2,804



ECOLOGICAL LAND UNITS

[1]Mapping unit complexes of two or more series were included within series of the first name of the complex for this tabulation.

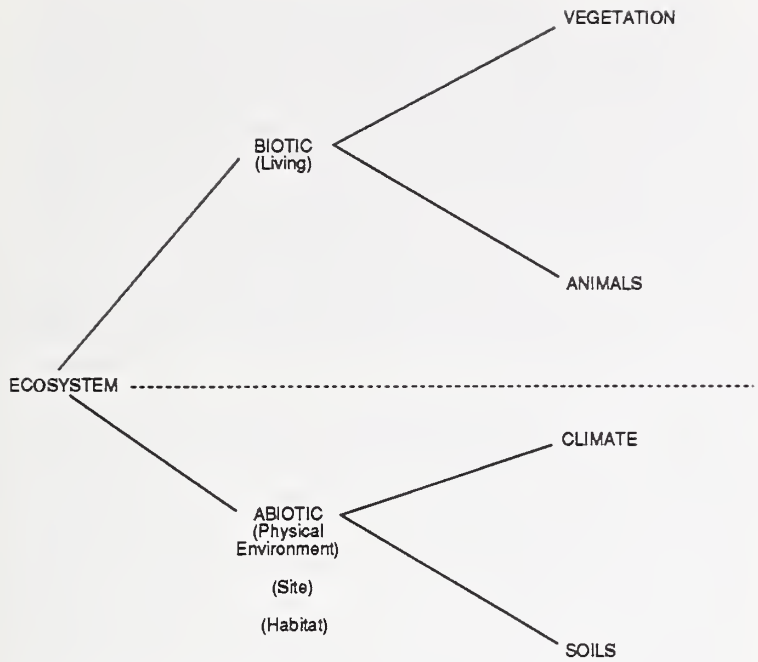


Figure 2—Ecosystem structure.

by many factors and difficult to predict with confidence. Our challenge over the years has been to describe, classify, and identify ecosystems to gain and communicate understanding of how they respond to management. One of the first steps is to illustrate the structure of ecosystems by separating and naming the components (fig. 2). Beyond this point is where most of us, through our education and experience, begin to diverge in perspectives.

As soon as you focus on one of the components, the others become secondary. For example, all soil scientists are well grounded in Jenny's basic equation:

SOILS = f(Climate, Relief, Parent Material, Vegetation, Animals, and Time)

In a parallel fashion, plant ecologists are well grounded in Major's basic equation:

VEGETATION = f(Flora, Climate, Topography, Soils, Animals, and Time)

Each of these equations represents the concept of an integrated expression of the ecosystem. However, a common mistake is to let the similarity in the functional form of these equations suggest that plant ecologists and soil scientists should be in complete agreement—or that soils and vegetation should be closely correlated.

As individuals we each see the ecosystem from one point of view—our own. If we do not communicate well it is not because one point of view is wrong and another is right. It is only because our perspectives are different. When developing plans for a building, the architect must present several perspectives before a whole picture evolves. To have meaningful dialog, we must make the attempt to see the situation from more than a single perspective. Before we can develop a shared perspective, we must understand each other's perspectives. The following perspectives outline how we think a soil scientist and a vegetation specialist view an ecosystem, and suggests an ecological perspective that would serve both better.

Soils Perspective

If my focus begins with the soil, I am interested in what kinds of soils exist on the landscape (taxonomy), how they are distributed (mapping), and why they are different. What is the relative magnitude of each determining factor? Can I sort them out one at a time and then put them together? What is the relationship of a specific soil to each of the soil forming factors? I would guess that parent material and climate are dominant and expect that differences among soils become greater with time. Animal and vegetation effects are often secondary. Soil taxonomy is based primarily on morphology, although climatic effects are specifically incorporated at the third and fourth taxonomic levels for moisture and temperature regimes.

Once I feel comfortable with understanding the soil, then I can begin looking at relationships with the vegetation and potential vegetation. Relationships of soils to landforms and certain vegetation types are useful aids for mapping. A better understanding of soil-vegetation relationships would be very useful.

Vegetation Perspective

If my focus begins with the vegetation, I am interested in what kinds of plant communities exist on the landscape (taxonomy), how they are distributed (mapping), and why they are different. What is the relative magnitude of each determining factor? Can I sort them out one at a time and then put them together? What is the relationship of a specific plant community to each of the vegetation development factors? Parent material and climate are usually dominant factors and changes with time and disturbance are dramatic. Animals and soil variables (especially parent material and hydric soils) are important in many areas but are usually secondary. Vegetation taxonomy is based primarily on flora composition and structure. If variation in time can be sorted out by identifying potential natural vegetation types, then these associations can be used as an integrated expression of the physical environment commonly called a habitat type.

Once I feel comfortable with understanding the vegetation, then I can begin looking at relationships with the soils and topography. Topographic and parent material relationships are very helpful for mapping the habitat types. A better understanding of soil-vegetation relationships would be very useful.

Ecologic Perspective

This perspective is needed by soil scientists to help move toward being soil ecologists, as well as by vegetation specialists, so they may operate as plant ecologists. The ecosystem perspective is available for all specialists if they choose to think from that perspective, and if their education and philosophy allow. Are you satisfied with

being soil scientists or would you rather be soil ecologists? Most of us have had some training in botany, soils, and ecology, but none has sufficient breadth and depth to be the ultimate expert in all three areas. We suggest that a sharing of talents and perspectives may provide the best hope for helping each other understand ecosystems and how to manage them.

The interrelationships of soil and vegetation have been a fascinating and frustrating area of investigation for many generations. Soil scientists and plant scientists have both championed their narrow, biased perspectives—generally from the point of view that "my perspective is better than yours." We are all limited by our perspectives, or in current terminology, our paradigms. We all have a natural degree of "paradigm paralysis," blinding us to new ideas and different perspectives. Fortunately, there are some "paradigm pioneers" willing to risk ridicule by examining and trying new ideas.

One of us remembers well an incident where John Arnold, a landtype mapping pioneer at the Boise National Forest, and Bob Pfister, a young habitat type enthusiast, were making separate presentations during a Forest Service Educators Tour in the Payette National Forest in 1970. After each waxed eloquent on the value of these independent approaches to land management planning, old John took the young upstart aside over a bottle of Jack Daniels and said, "If we don't point out to all these administrators that both of our approaches have unique and complementary value, the chiefs will say, 'Why do we need two approaches?' and will proceed to support one and throw the other one out." The next day we made a joint presentation, and the next week we were cooperating on a resource inventory of the Idaho Primitive Area. We did not call it that, but, in effect, we were working together to gain a shared ecosystem perspective.

The Society for Range Management has been actively debating soil-vegetation relationships and terminology during the 1980's. Although many individuals still do not agree, the framework is being laid for effective communication. The range site/habitat-type arguments may not be resolved until there is a one-generation turnover, but students of the subject will be approaching the subject with a broader paradigm than their predecessors. The principle of factor interaction and compensation makes it impossible for a habitat type to equal a range site if the former is based primarily on similar vegetation potential and the latter is based primarily on similar physical site characteristics. As John Arnold used to frequently remind people, "Wishing won't make it happen and saying it doesn't make it so."

ECOSYSTEM CLASSIFICATION AND MAPPING

Although many individuals have worked and are working hard to develop and demonstrate holistic perspectives for ecosystem classification and mapping, the universal solution has not yet been established. However, some benchmarks can be noted toward this end.

One example of a Forest Service mapping program was the Landtype Inventory in the Intermountain and Northern Regions during the 1970's (Wertz and Arnold 1972). These maps were based primarily on recognition of similar landform descriptions of major soils and habitat types as the source of management interpretations. They were especially useful for the ecological approach to land use planning that characterized the 1970's planning efforts. These maps probably should be dusted off and used as a layer in your developing Geographic Information Systems to help address current concerns dealing with landscape ecology.

In 1970 the Forest Service commissioned a task force to develop a single, hierarchical classification of ecosystems from existing information. The task force realized that responding fully to such a request was impossible, but suggested a methodology to meet the expressed objectives (Pfister and others 1972). This was approached from a mapping perspective where habitat type mapping and landtype mapping could be combined at different levels in their respective hierarchies to define an Ecological Land Unit, an area of land with defined similar vegetation potential and soil/landform characteristics. This attempt to reach an integrated ecosystem perspective was used by several people for planning in the 1970's with some modifications. The landtype mapping experts were reluctant to support the concept because they thought their landtype approach already did this. However, independent maps of landtype and landtype phase compared with habitat type maps suggested that theory and reality were not necessarily the same.

A decade later another task force focused on the taxonomy used to classify inventory points and provided the term Ecological Response Unit for any level of integration from potential vegetation types and soil taxonomy hierarchies (Driscoll and others 1984). The focus in this effort was on standardization of taxonomy for point identification. This provided a means to aggregate, but the link to mapping for disaggregation was not clearly established. In spite of five agencies approving the standardization, acceptance and use have not been automatic.

A Range Standardization Committee in the Society for Range Management has made good progress improving concepts and terminology. One or two major problems remain, but the desire and dedication are commendable. New terms have been defined to resolve ambiguities of older terms, but this does not necessarily solve the basic problems. A few of the definitions in the Society for Range Management (1989) glossary still represent unique perspectives within the profession.

The California National Forest System (Allen 1987) provides a good starting point for a regional program developed in the middle of widely varying viewpoints. The ecological type is basically a subdivision of a habitat type, where needed to reflect meaningful differences in soils or productivity within a habitat type.

We are probably all in complete conceptual agreement at the lowest level in any of these classifications. We could stand on the ground and agree on the smallest area

that has similar vegetation potential, soils, and topography, and hence (we would hope) similar productivity potentials, hazards, and responses for management. However, if classifications are useful only because they simplify to a number of categories that we can work with, then the ultimate question becomes one of scale and practicality. Within a State we would expect to have 30 to 80 forest habitat types and a similar number of soil series on forested lands. However, at the same level of ecological response units, ecological sites, or ecological types, as currently defined, we would expect 100 to 300 taxonomic units.

An alternative to taxonomy or mapping to the *n*th degree is to use multiple classifications that already exist to create unique units as needed. The National Land Classification experience offers the Ecological Response Unit concept for sorting, aggregating, and analyzing inventory and other data bases. The ECOCLASS experience offers the Ecological Land Unit concept for mapping, landscape ecology, and management. With the advent of GIS, this will be straightforward, objective, rapid, and efficient. Decisions must be based on appropriate levels of implementation (scale and costs) relative to objectives.

These ecosystem classification examples illustrate the difficulty of dealing with problems that are larger than a single profession. One approach is to try to expand the profession to cover the scope of the problem. However, the more logical approach may be for the professions to pool their collective expertise in teamwork endeavors for certain problems. The natural resource ecological issues of the 1990's will require a higher level of mutual support among agencies and professionals than ever before to maintain or regain leadership roles. This will require setting aside professional arrogance, agency arrogance, and the NIH (not invented here) syndrome. All we have is each other and a professional responsibility to manage natural resources for a sustainable society.

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Speakers answered questions from the audience after their presentations. Following are the questions and answers on this topic:

Q. With soil/vegetation interactions most prominent at forest-grassland interfaces, would the soil/vegetation interactions also be prominent at the geographical limits (particularly dry end) of individual species of conifers or understory vegetation?

A.—Yes, species are often restricted to certain substrates near their dry geographical limits. This is particularly evident in the Wind River and Bighorn Ranges of Wyoming where many northern Rocky Mountain species are near their eastern limits at those latitudes.

THE RELATIONSHIP BETWEEN SOILS AND VEGETATION

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ABSTRACT

Numerous studies have failed to show consistent relationship between soils and vegetation. A review of basic concepts of soil and vegetation development indicates that vegetation and soils are mutually associated with each other, both being the product of the same environmental variables. The mutual association is between the whole soil and vegetation, not individual soil properties and vegetation or selected attributes of vegetation. At any scale beyond a very localized application, a universal correlation does not exist between soil properties and vegetation attributes. Even within very restricted geographic bounds, variation between attributes can yield untenable results. Soil bodies, when grouped and classified at the soil series level, should correlate reasonably well with habitat type. Conceptually, each habitat type is associated with a unique set of soil series.

INTRODUCTION

Numerous studies have attempted to correlate climax vegetation and soils (Daubenmire 1970; Daubenmire and Daubenmire 1968; Jensen and others 1990; Neiman 1988; Sexton 1986; Tisdale and Bramble-Brodahl 1983) and, in general, the results have been disappointing. There have been few successes demonstrating correlation between selected soil properties and vegetation. Using factor analysis and multiple discriminant functions, Bunting (1978) was successful in correlating selected soil properties with major vegetation types in the Guadaloupe Mountains in southwestern Texas. Working in an area where parent materials were similar, Tisdale and Bramble-Brodahl (1983) showed site variables, including soil pedon properties, discriminated between habitat types to a fair degree. Klemmedson (1964), working with seral vegetation in a very localized situation, demonstrated the relationship between site factors and vegetation.

In general, studies involving soils derived from different parent materials have done poorly (Neiman 1988; Sexton 1986). Attempts to predict tree site index using selected soils and site properties have been inconsistent (Copeland 1958; Monserud 1990).

The numerous failures suggest that a basic soil/vegetation relationship does not exist or something fundamental has been overlooked. The sporadic successes in correlating soils with vegetation appear to be locally significant, but are not applicable elsewhere. Johnson and Simon (1987) had some success in broad relational correlations; for example, *Festuca idahoensis* occurred on loess soils, whereas *Agropyron spicatum* did not, *Pseudotsuga menziesii* and *Pinus ponderosa* occupied soils without Mazama ash, and *Abies grandis* and *Abies lasiocarpa* were associated with soils without ash. Although supported with data from 775 intensively sampled plots, they were unable to extrapolate their correlations between soils and vegetation outside their area of study. The high number of failures that have been published is surely dwarfed by the number of studies that were not reported. A review of some of the basic concepts appears in order.

FUNDAMENTAL SOIL EQUATION

Let us start with Jenny's (1941, 1958) widely accepted basic soil formation equation:

Soil = f(climate, parent material, relief, organisms, time)

This tells us that if the soil pedons at two different points on the landscape have, in all details, the same kinds and intensities of soil properties, the soils of the two places are the same and they have had the same developmental history. Conceptually, the productivity of the soil at the two places also would be the same, because the physical and chemical properties of the soil at the two places would be the same. And, should the soil pedons be classified according to conventional soil classification standards, they would be classed as members of the same soil series, the basic unit of soil classification.

VEGETATION-SOIL DEVELOPMENT

A less familiar concept is that each soil body is associated with a specific climax vegetation. This is supported by Major's (1951) deduction and rationale that the same environmental factors responsible for soil formation are also responsible for the vegetation that is produced. The same independent variables involved in the soil equation are included in his equation on plant community:

$$\text{Vegetation} = f(cl, pm, r, o, t)$$

A particular climax vegetation as well as a specific soil would result under a fixed set of independent variables. Vegetation, like soil, is the product of the same group of

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independent variables. Vegetation and soil mutually influence each other and neither is the result of the other (Jenny 1958; Major 1951).

The two fundamental equations show that a soil body is associated with a specific plant community, the climax plant community. This is an important point. This means that all points on the landscape with the same soil are associated with the same climax plant community, the same plant association, and the same habitat type. The plant association is the basic classification unit of climax vegetation and the habitat type is the land classification unit that supports or supported a particular plant association (Daubenmire and Daubenmire 1968).

Different soils are known to support the same climax vegetation (Daubenmire 1979; Daubenmire and Daubenmire 1968; Hironaka and others 1983; Neiman 1988). The fact that the same aggregate of plants can grow and thrive on different soils means the same climax vegetation can be supported on more than one soil. This phenomenon is explained by factor compensation of plants. Plants are able to grow and thrive over a range of conditions, permitting the same plant community to occur on different soils.

SOILS AND HABITAT TYPE

The relationship between soils and habitat type has not been easy to demonstrate because of fuzziness of how much variation we include in our abstract classification units of soil and vegetation. Conceptually, the units can be defined, but the problem is whether they are sufficiently close to ecological reality. Because of the lack of undisturbed vegetation, we are unable to confirm whether our assumptions about vegetation are correct.

Assuming we are correct, each soil series is associated with a particular climax community, and in turn with a particular habitat type. Otherwise, Major's (1951) vegetation equation is flawed. Since it is likely that more than one soil series is associated with the same climax community, by definition then, each habitat type is associated with a unique climax vegetation and a unique set of soils. It also tells us that all soils included in a soil series are associated with a single habitat type and no other. Should this not be the case, the soil or the habitat type is misclassified.

Not only should all soils of a soil series support the same climax community, they should also have the same productivity. It must be kept in mind that soils of different soil series may possess the same productivity. While differences in pedon properties or intensities justify placing soils in different soil series, the differences may not affect productivity because of plant compensation. That is to say, soils of different soil series may possess the same level of productivity.

SOILS AND RANGE SITE

The concept that soil series and productivity go together in wildland management is the basis of the range site classification (Shiftlet 1973). Here each unit includes soils of several soil series of similar productivity that support or supported the same climax vegetation. For those familiar with habitat type classification, the range site classification would be comparable to the habitat type phase, if the phase were to be based on productivity (Hironaka 1986). If one were to group all range sites capable of supporting the same climax vegetation, the group of range sites would be equivalent to a habitat type (Hironaka 1986). Therefore, range sites represent differences in productivity within a habitat type. Unfortunately, the nomenclature of the range site classification does not identify or allude to the climax vegetation, and grouping of range sites into habitat types cannot be easily done.

FOREST SITE PRODUCTIVITY

The site index has been a primary means of assessing forest site productivity. In essence, the average height of dominant trees at age 50 or 100 years is used as the site's index of productivity. Site index curves have been developed for various timber species for local and regional interpolation. To estimate site index from soil and site characteristics, attempts have been made to correlate known index values with site and soil parameters to develop prediction equations. Recently, using stem analysis to get direct measure of site index of inland Douglas-fir, Monserud (1984) found that site index for old inland Douglas-fir tended to drop off considerably because the older measured trees occurred on "poorer" and more inaccessible sites. This suggested that the older trees occurred on different and less productive soils. In a follow-up study Monserud and others (1990) attempted to correlate site and soil factors with site index by partitioning the habitat type series. They came to the overall conclusion that site index correlations with site and soil variables were not very reliable.

Jenny's (1941) contention that no soil property can be universally correlated with property of vegetation and Major's (1951) stress that mathematically there are no universal correlations between vegetation and soil properties as neither is determined by the other have been ignored. In a nutshell, this is the basic reason why seeking of mathematical correlation of soil properties and vegetation has failed in the majority of cases. Only with studies where the independent variables have been sufficiently constrained have meaningful correlations been demonstrated. It would appear that the forest site index would be closely predicted by the soil series. It is something to think about.

SOIL CLASSIFICATION AND FOREST AND RANGELAND MANAGEMENT

The practice of classifying and mapping soils at the soil family level of classification does not work very well for intensive land management. This is especially true when it is not used in conjunction with the habitat type classification. The soil family class groups soil series on the basis of broad textural, mineralogical, and soil climate similarities. This level of classification often includes soils of different habitat types and productivity levels, and greatly weakens the reliability and effectiveness of management prescriptions.

For intensive management of forests and rangelands, a baseline soil map at the soil series level of classification is something to work toward. Used in conjunction with habitat type, community type, and topographic maps (GIS overlays), much could be known about any piece of land before leaving the office. Supported by an information storage/retrieval system based on habitat type, community type, and soil series and soil series phase, much detailed management information could be made available concerning any piece of inventoried landscape. More important, it would provide a means whereby new information relating to land treatment and responses can be inputted and retrieved for later use by the next generation of land managers. The new generation of land managers need not reinvent the wheel as we have repeatedly done in the past.

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INFLUENCE OF FIRE ON FACTORS THAT AFFECT SITE PRODUCTIVITY

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ABSTRACT

Presettlement fire played an important role in nutrient cycling, plant succession, diversity, and stand dynamics in coniferous forests of western North America. Prescribed fire can maintain site quality and contribute to control of insect and disease problems while reducing wildfire hazard. Fire effects on soils are largely governed by interactions between fuel consumption and soil characteristics that influence soil heating. Many impacts on vegetation and site productivity are also related to soil heating.

INTRODUCTION

Before the coming of western civilization, fire played an important role in nutrient dynamics, in plant succession, and in shaping stand density of coniferous forests in western North America (Arno 1980). Lightning was a common source of ignition, and Native Americans commonly used fire to manipulate grasslands and forests (Barrett and Arno 1982; Gruell 1985). However, Europeans who settled the West viewed fire as a destructive agent (Pyne 1982). The fires of 1910 in the Northern Rocky Mountain region greatly influenced Forest Service fire policy (Pyne 1982), and resulted in an obsession to suppress all fires. Beneficial uses of fire were largely ignored until the 1970's (Pyne 1982).

Concepts of multiple use and sustained yield are traditions in American forestry; but concerns about wildlife, threatened and endangered species, global change, and forest health have led to new demands on forests. The "new perspectives in forestry" concept raises additional questions about long-term productivity of forests. The total amount of plant material produced by a forest per unit area per year (net primary productivity) is a measure of forest productivity that integrates environmental site conditions, vegetation, and stage of vegetation development (Grier and others 1989). A better understanding of site productivity is needed to assess the effects of management activities, including fire's effect on factors that influence productivity.

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It is clear that soil is a major determinant of site productivity and environmental site conditions are both extrinsic and intrinsic (Grier and others 1989). Extrinsic site factors (climate, topography, and geology) determine long-term regional productivity potentials and set the upper limit of site productivity. Intrinsic factors such as soil moisture-holding capacity, soil nutrient status, and soil porosity affect plant growth and productivity potential. Natural disturbances such as fire, insects, disease, and erosion alter intrinsic factors, influencing soil forming processes, vegetation development, and site productivity. Potentially, vegetation development and site productivity can be influenced by the use of fire, timber harvesting, and chemical application.

The purpose of this paper is to review the interactions between fire, vegetation, and soils that influence site productivity. We will first examine historic fire occurrence: frequency, size, and characteristics. Next, we will discuss variations in heat output from fire and how it affects soil heating and in turn, intrinsic site factors. Finally, we will discuss how fire affects the physical, chemical, and biological systems of the soil.

HISTORIC FIRE OCCURRENCE

Study of historic fire frequencies, sizes, and effects provides insights into the ecological role of fire and how it influences forest productivity. Since fire is a natural part of the development of coniferous forests in western North America, the productive nature of the ecosystems is likely adapted to fire. There is considerable variation in relationships between fire occurrence, site productivity, and fuel accumulation. Decomposition and nutrient cycling processes in some ecosystems were controlled by recurring fire. In other ecosystems fire was relatively infrequent and herbivory and fungi assumed greater roles in decomposition and nutrient cycling (Gosz 1981; Olson 1981; Woodmansee and Wallach 1981). At the landscape level, fire potential can be generalized on the basis of productivity and physiognomy (fig. 1). Sites that are severely moisture or temperature limited are unproductive. These sites were fuel limited, and burned infrequently in the past (Martin 1982; Ryan in press). Such sites were dominated by herbaceous plants or shrubs, but few trees. Higher temperature and moisture in mesic coniferous forests increased productivity and fire frequency. In highly productive, but more moist ecosystems, large amounts of burnable biomass accumulated, but meteorological conditions were seldom favorable for extensive burning.

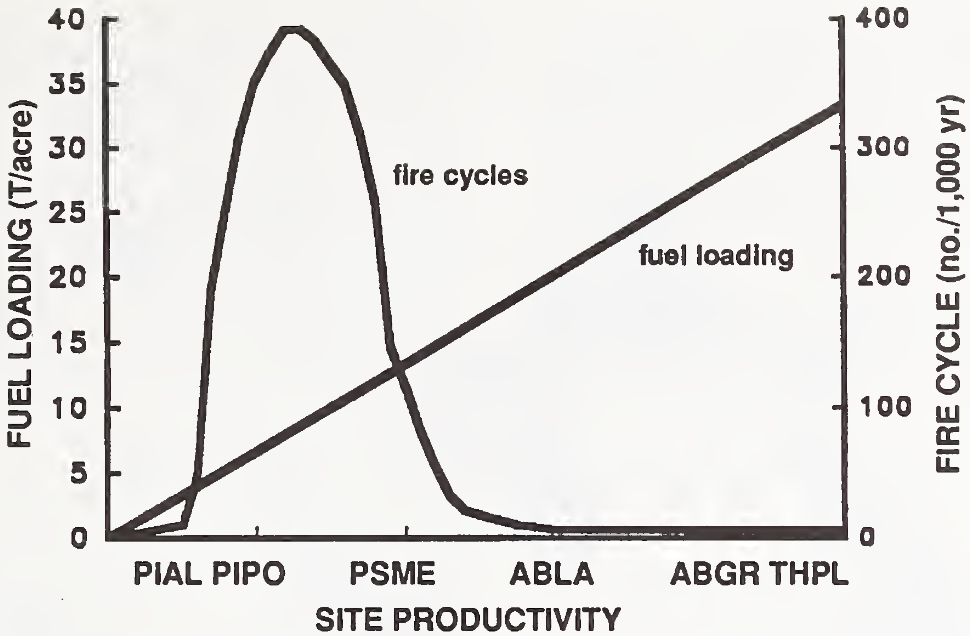

Figure 1—General relationship between site productivity, fuel loading, and fire cycles in the Intermountain West. Site productivity versus habitat type trends are excerpted from Pfister and others (1977). Fuel loading versus productivity trends are adapted from Brown and See (1981). Fire cycles were adapted from Martin (1982).

Biomass increases with time until it is removed by fire, logging, or grazing, or until litter accumulation and decomposition are in equilibrium (Gosz 1981; Olson 1981). Fire frequency increases as aboveground biomass approaches some lower limit capable of sustaining a fire (Martin 1982; Olson 1981). In western coniferous forests, moisture deficits vary from chronic on arid and less productive sites to infrequent on the most mesic sites. However, even on the most mesic sites, moisture deficits were frequent enough that fires likely consumed live and dead biomass before litter accumulation and decomposition were in equilibrium.

In low-elevation forests, light surface fires routinely spread from rangelands and seral grasslands. Fires burned through cured herbaceous fuels as often as every other year (Martin 1982), but fire frequencies of 5 to 20 years were more typical (Arno 1980; Martin 1982; Steele and others 1986). Fires often top killed shrubs and reduced encroachment of young conifers into rangelands (Gruell 1986; Gruell and others 1986), but they rarely killed the scattered large conifers. Such fires produced minimal soil heating (Ryan and Noste 1985). Frequent fires maintained open "parklike" stands of ponderosa pine (*Pinus ponderosa*) with well-developed herbaceous understories. Allocation of site water and nutrients among a few widely spaced trees resulted in vigorous growth and, apparently, relatively high resistance to insects and disease (Christiansen and others 1987; Crookston and Stark 1985).

During the early settlement of the West, extensive grazing removed much of the herbaceous fine fuel in lower elevation forests (Gruell 1986). These forests were easily accessible and extensively logged. Logging and grazing altered the role of fire as an ecosystem process. In the absence of periodic surface fire these forests accumulated fuels (Brown 1985; Ryan and Frandsen in press) and became overstocked with dense thickets of young trees and shrubs (Arno 1988; Gruell 1986; Gruell and others 1982). These forests frequently experienced severe insect and disease problems that caused mortality and created fuels.

In the lower montane zone, fires burned on the average of every 20 to 50 years (Arno 1980; Martin 1982). At higher elevations fire frequencies of 50 to 100 years were more common. Surface fires commonly burned beneath stands, injuring and killing some trees and igniting smoldering ground fires (Ryan and Frandsen in press). Soil heating was variable as a result of spatial variation in ground and surface fuels. These surface fires released nutrients and reduced competition from late-successional and climax species. Stand replacing crown fires occurred during dry years where fuels had accumulated, resulting in patches of nonforest or young forests across the landscape.

In the subalpine forests, presettlement fires were less frequent than at lower elevations. On productive sites, the fire regime intergraded with that of the montane forests. On less productive, drier, or colder sites, the minimum time to accumulate enough dead biomass to carry a fire in the absence of high wind was 20 to 50 years. Fires generally burned surface herbaceous fuels, frequently torching clumps of trees. Even light surface fires resulted in high mortality of the typically fire sensitive subalpine conifers (Ryan and Reinhardt 1988). Fire suppression in subalpine forests may not have had a major effect on fuels, species composition, or stand structure; but it probably affected the age distribution of subalpine forests.

The prevalence of fire on the landscape was repeatedly observed by 19th century explorers. Evidence of frequent large fires was observed by Ayres (1901) in the Lewis and Clark Forest Reserve of Montana (includes the Bob Marshall and Scapegoat Wilderness complex) and by Leiberg (1899) in the Bitterroot Forest Reserve. Nearly a million acres (31 percent of the area) burned in the Lewis and Clark

Reserve (fig. 2) between 1859 and 1898 (Ayres 1901). The 1889 fire in the Lewis and Clark Reserve burned about 800,000 acres in a stand-replacement fire. Leiberg estimated that 40 percent of the Bitterroot Reserve burned during the same period. Barrett (1982) found that some areas of Cook Mountain, in the Clearwater National Forest in Idaho, burned as many as five times between 1869 and 1919.

Some areas of the 1889 fire reburned in 1895, 1910, and again in the 1988 Canyon Creek fire (250,000 acres). Fires similar in size to the Canyon Creek fire historically occurred on the average of every 25 years in this area (Losensky 1990).

Many of the large fires of the 19th and early 20th century consumed all the forest floor, leaving the soil surface bare. These observations suggest that, historically, the burn interval was short enough to preclude the accumulation of forest floor depths typical of what we see today. Large areas of the forest were apparently maintained in earlier stages of succession than what we typically see today (Gruell 1983; Losensky 1990) (fig. 3). The effects of fire on site productivity remain uncertain, but it is apparent that fire played a role in nutrient cycling, species composition, and stocking control in virtually all western coniferous forests.

FIRE CHARACTERISTICS

Fire characteristics vary with fuels, weather, and topography. Some fires exhibit flaming, others smolder. Flames associated with forest fires are the result of the oxidation of pyrolysis products released from heated materials. Smoldering is pyrolysis and oxidation without flames. Crown fires, surface fires, and smoldering ground fires have different heat release and spread rate characteristics. Each fire type results in different levels of fuel consumption and above- and belowground heating. Surface fires are confined to litter, twigs, branches, and larger woody fuels on the forest floor. Live vegetation may add to the surface fire depending on its flammability. Fine surface fuels make up a small portion of the fuel loading on most sites, but their ignition and consumption determine fire spread, intensity, and ignition of other fuels. Rate of fire spread is primarily dependent on the moisture content of the fine fuels and wind speed. A light surface fire passing through needles, twigs, and small branches will not significantly heat either the crown foliage or mineral soil but will dry surface fuels. Surface fuels, twigs, cones, and logs act as ignition sources for the forest floor. Larger fuels are of less importance to fire spread, but contribute heat for the initiation of crown fires, drying of fuels, and the downward flow of heat into the mineral soil.

Flame length is a visual measure (Byram 1959) of fire intensity, which is the rate of energy released by the fire. If fire intensity is sufficiently high, flames may reach into the canopy and ignite the tree crown. Less intense surface fires may ignite understory trees and dead branches in the lower crown and carry the flames into the crown. Crowning and torching result in rapid spread rates and intense heat release for short durations, which results in little soil heating. Smoldering fires have low intensity and occur in tightly packed fuels of low moisture content. The tightly packed fuels on the forest floor are the fermentation (F) and humus (H) layers, which are made up of partly to well-decomposed organic matter. The F and H layers are roughly equivalent to the Oe and Oa horizons defined by the Soil Conservation Service, U.S. Department of Agriculture, (Federer 1982). In this paper, we will refer to the F and H layers combined as duff. The litter (L) layer is not part of the duff. Smoldering occurs in the forest floor duff (F and H layers) where packing ratios (volume fraction of fuel) are greater than 10 percent (Frandsen 1989). In contrast, surface litter packing ratios are less than 10 percent and exhibit flaming.

The term fire severity has been used to describe fire effects based on the appearance of the litter, duff, and soil after burning (Wells and others 1979). Severity reflects the degree of organic matter consumption and soil heating (Morgan and Neuenschwander 1988; Ryan and Noste 1985), but is ambiguous in quantification. In many instances, the term fire severity has been used synonymously with fire intensity. Present usage is confusing. In this paper we will discuss soil heating and the relationships to soil chemical, physical, and biological properties.

Heat flux describes the rate of heat flow that is delivered to the plant or the soil horizon of interest. The duration of heat flow and the resultant temperatures directly affect the soil-vegetation system (Wells and others 1979). Since we know much about the chemical, physical, and biological responses to specific temperatures, it is useful to express heat flow as temperature profiles when explaining fire effects (Harvey and others 1989).

SOIL HEATING, TEMPERATURE, AND TRANSFER

Rate of burning and heat output for surface and crown fires are influenced by fuel load, fuel moisture, topography, ignition method, air temperature, wind, and relative humidity. Heat output and burn rate for smoldering are influenced, in part, by the fuel moisture and inorganic content of the duff (Frandsen 1989). Rates of spread for flaming are a thousand times greater than for smoldering. When duff moisture content is high, heat from the surface fire may dry, ignite, and burn a portion of the duff (Ottmar and others 1985; Sandberg 1980), but only minimal soil heating results. When dry duff is present, smoldering is the primary source of heat (except for large fuel concentrations) to the lower duff and mineral soil (fig. 4). Although temperatures of smoldering duff (500 to 600 °C) are lower than for flaming (1000 to 1500 °C), the long duration of smoldering and the close proximity of duff to the soil results in greater heating. When duff does not burn, it becomes a barrier to heat flow to the mineral soil (Frandsen and Ryan 1986).

Attempts to understand soil heating by characterizing fire intensity have not been successful. Fire treatments have been characterized according to fire intensity (reaction intensity and fire line intensity) (Chandler and others 1983), visual estimates (Ryan and Noste 1985; Tarrant 1956a), heat fluxes, and amount and rate of fuel consumption (Albini 1975). A major problem with predicting soil heating is the inability to quantitatively relate prefire conditions and fire characteristics to the downward heat pulse.

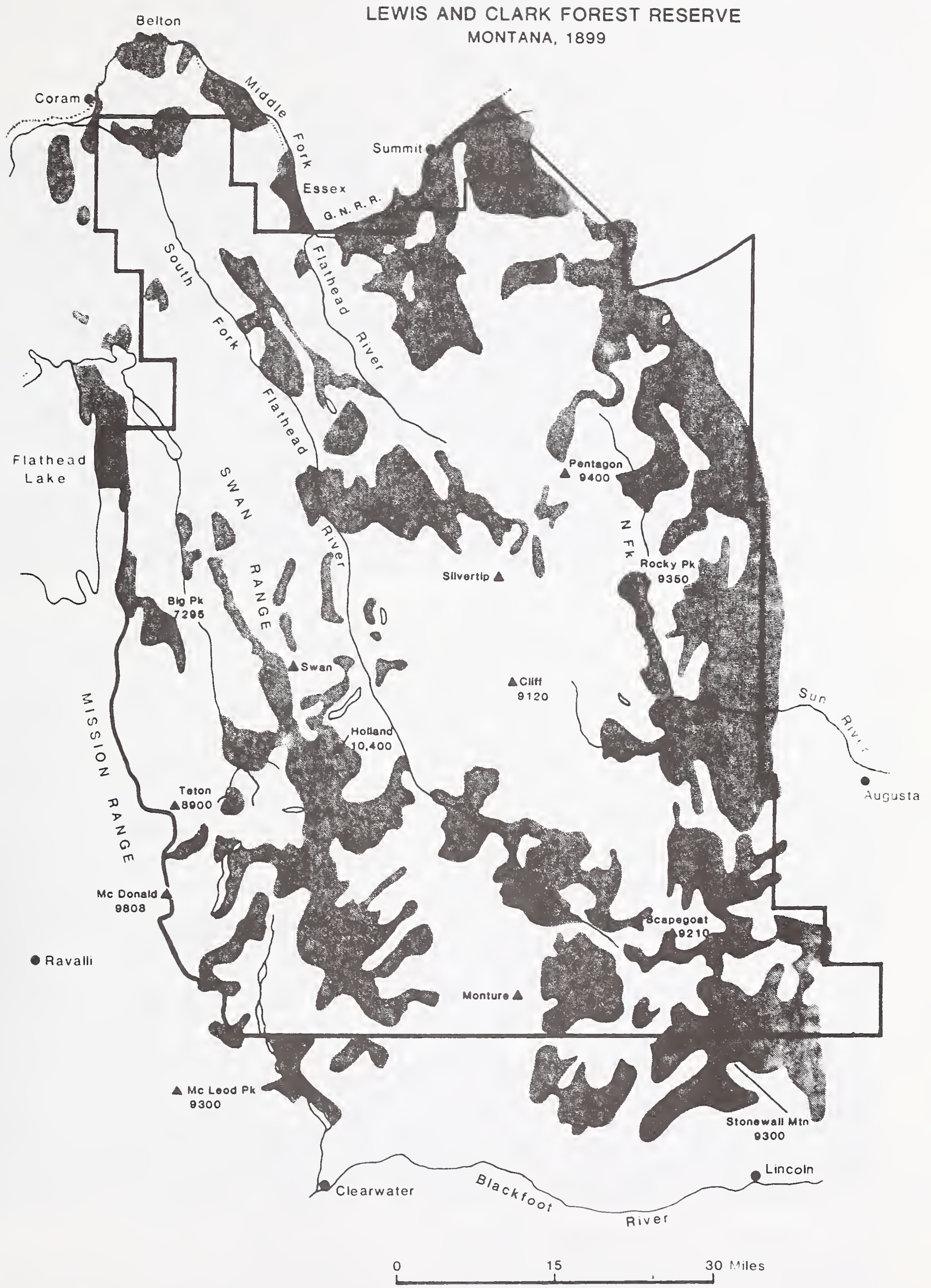

Figure 2—Areas burned in the 40-year period from 1860 to 1899 in the Lewis and Clark Forest Reserve. The shaded areas represent stand-replacement fires. Areas that underburned are not indicated. Map is redrawn from Ayres (1901).

A

B

Figure 3—This photo comparison contrasts the appearance of a forest in 1900 (A) and in 1981 (B). The photos were taken from a ridge about 5 miles west of Haystack Butte, looking southwest across Smith Creek toward Crown Mountain on the east front of the Rocky Mountains, Lewis and Clark National Forest. In (A) the near slopes are in early succession following wildfire in the late 1800's that removed conifers and stimulated shrubs. In (B) the near slope in 1981 is now densely covered by Douglas-fir. Photos are taken from Gruell (1983): (A) is USGS photograph 665 by C. D. Walcott, and (B) is a photograph by G. E. Gruell.

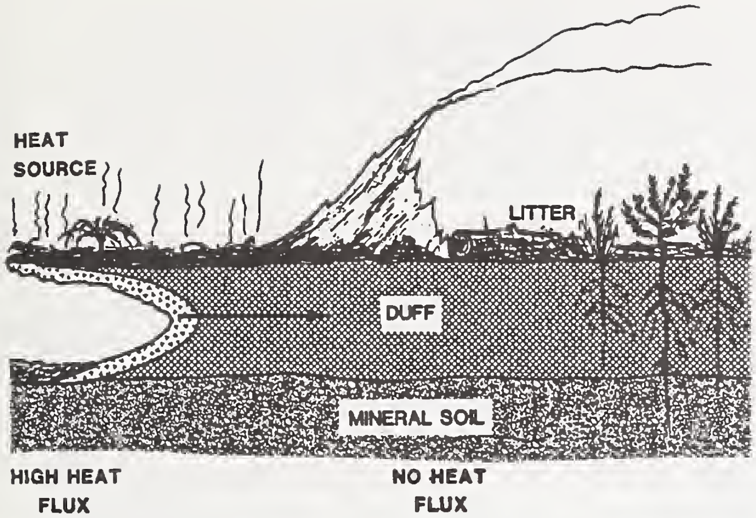

Figure 4—Diagram showing heat flux to the mineral soil where duff burning is the heat source. When duff does not burn, it is a barrier to heat flow. The arrow represents the direction of spread for duff burning at a rate of about 3 cm/hour.

Rowe (1983) noted that release of energy as the fire passes may bear little relation to degree of soil heating as indicated by survival of plant parts buried in the soil. Shearer (1975) also showed that fire intensity alone was not closely related to soil heating as indicated by root mortality. Moisture content of the duff and mineral soil and duration of the fire were also important. Ryan and Noste (1985) stated that factors other than fire intensity determine the downward heat pulse. Models such as the duff consumption model (Van Wagner 1972) and the duff burnout model (Albini 1975) have not been useful for predicting slash burn heat release (Wells and others 1979). The formulation of a purely physical model describing downward heat flux to the litter and soil would be valuable (Wells and others 1979).

Maximum soil temperatures and duration of heating (fig. 5) varied between grass, shrub, and forest fires where duff burned (DeBano and others 1979; Frandsen 1989; Scotter 1970; Wells and others 1979). Soil heating and temperatures also varied within fires (DeBano and others 1979; Shearer 1975; Wells and others 1979). DeBano and others (1979) constructed stylized curves to predict soil temperatures for light, moderate, and intense chaparral fires. Models based on heat-transfer theory (Bristow and others 1986; DeVries 1958; Wierenga and DeWitt 1970) were developed to predict heat flow. Scotter (1970) attempted to model heat transfer and predict soil temperatures beneath fires, but did not include moisture-aided heat transfer. Aston and Gill (1976), Campbell and others (1990), Chinanzvavana and others (1986), and Pafford and others (1985) modeled the downward heat pulse in soil beneath fires. Chinanzvavana and others (1986), Pafford (1986), and Pafford and others (1985) evaluated heat-transfer processes between the fire and the underlying soil surfaces. These studies provide useful insights into the process of soil heating, but the models remain untested and cannot be used to predict the effect of fire on site productivity.

Heat flow from a fire through the soil-biota system is illustrated (fig. 6) by a conceptual model (Hungerford 1990).

Heat generated from surface or ground fires is lost upward, transferred downward to the soil, or transferred downward to the duff if duff is not involved in the burn. Models of heat flow across the boundary layer between a fire and the soil are complex because of abrupt and dynamic changes in factors affecting heat transfer. Heat from the fire is transferred to unburned fuels, the atmosphere (air in fig. 6), and the soil or litter surface by radiation, convection, and conduction. Convection and radiation dominate, while conduction is important within a fuel element and between elements in contact. Only 8 to 10 percent (maximum of 25 percent) of the heat from burning is transmitted downward in the soil or litter (DeBano and others 1977; Packham 1969; Raison and others 1986; Steward 1989). The remaining 75 plus percent is lost upward and affects aboveground biological systems.

Heat transfer by conduction is important, but at soil temperature gradients created by fire, vapor flux accounts for the majority of heat transfer (Campbell and others 1990) and was as high as 60 percent (Westcot and Wierenga 1974). Liquid flux probably contributes little (10 percent or less) to heat transfer. Heat-transfer rates are influenced by numerous variables (fig. 7). Soil properties such as porosity, mineral content, water content, and organic content influence thermal conductivity, heat capacity, vapor conductivity, and hydraulic conductivity. It is clear from examination of the factors affecting heat transfer that knowledge of more than the fire characteristics is necessary for predicting soil heating and fire effects.

Little is known about heat transfer in living tissue. Most articles discuss heat transfer between the atmosphere and aboveground plants, particularly leaves (Campbell 1977; Gates 1980; Raschke 1960), or transfer within the stems of trees (Derby and Gates 1966; Herrington 1969; Martin 1963). Many of the variables that influence heat transfer in the soil (fig. 7) also influence heat transfer within living structures. The size (surface to volume ratio) and character of the biological structure affect heat capacity and thermal conductivity, thus influencing the final temperature. Thermal equilibrium with the soil likely will be established rather quickly for small structures, such as fungi and other

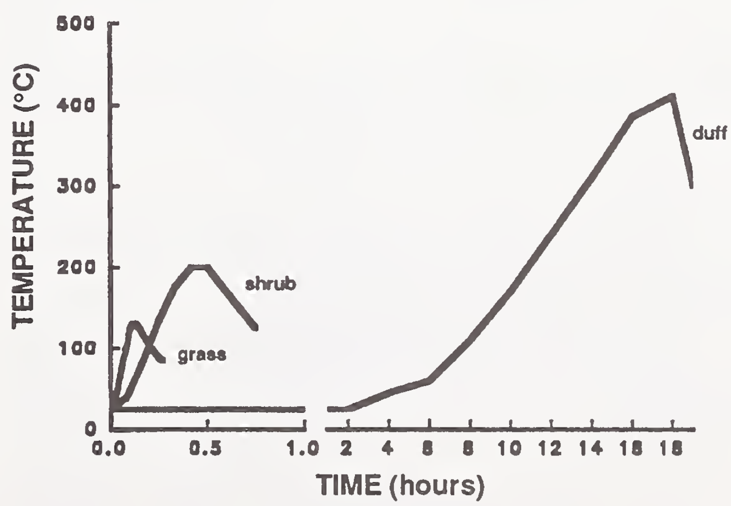

Figure 5—Maximum temperatures in the upper 2 cm of the mineral soil under a grass fire (Scotter 1970), a shrub fire (DeBano and others 1979), and a duff fire (Frandsen 1990).

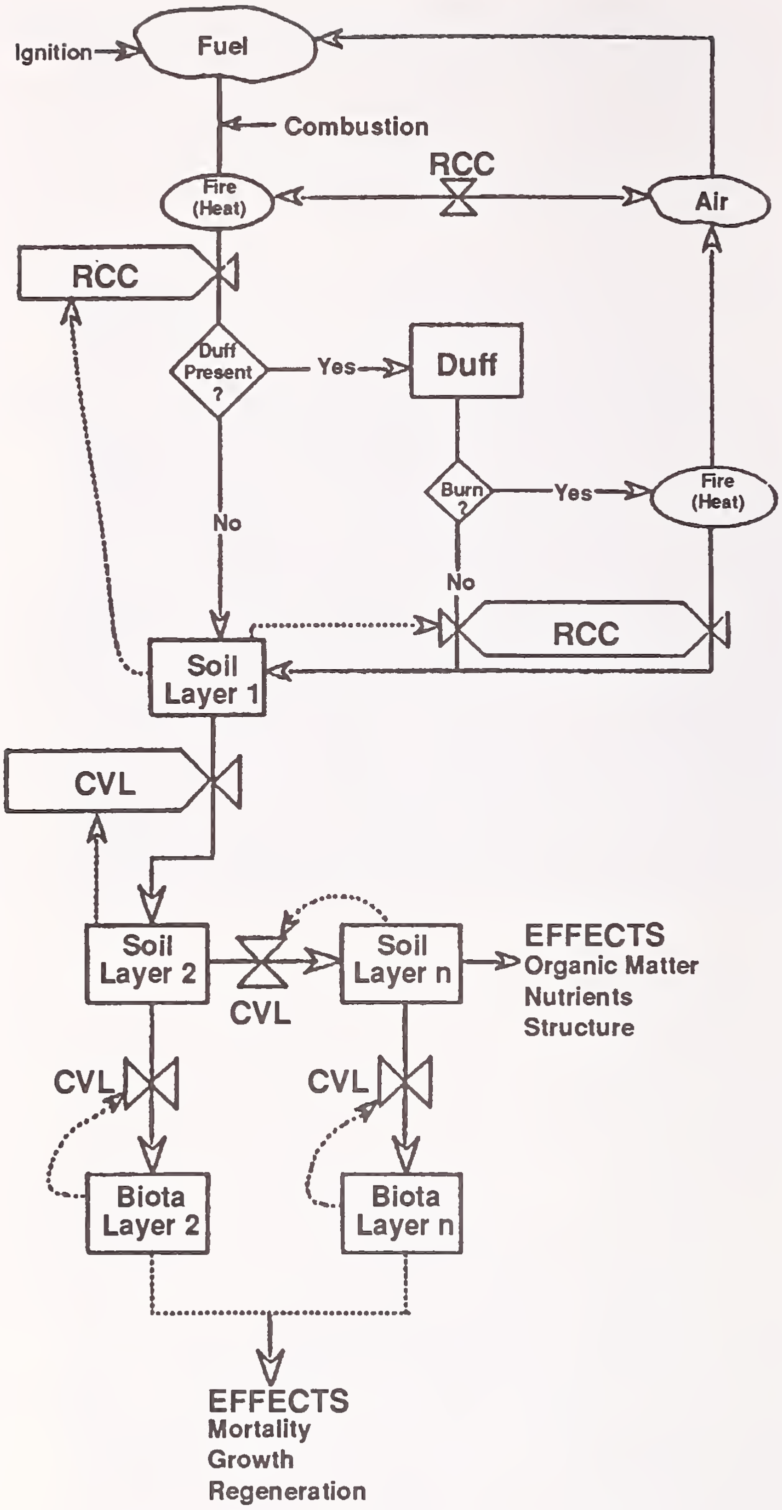

Figure 6—Model diagram of downward heat pulse from fire into the soil and biological systems. Rectangular boxes represent variables and solid lines between the boxes represent heat flow between the boxes. The processes that control heat flow between the boxes are shown by the "bow ties" on the solid lines. The dotted lines to the "bow ties" represent factors of the variables that influence the processes and rates of heat transfer between the boxes. RCC = Heat transfer by radiation, conduction, convection; CVL = Heat transfer by conduction, vapor, liquid.

microorganisms, and small plant parts. For larger structures such as roots, large rhizomes, tubers, and root crowns, temperatures may be different from the soil; thus heat transfer analysis is needed to evaluate the effects of soil heating.

EFFECTS ON FUEL CONSUMPTION

Fire effects on the physical, chemical, and biological components of the duff and litter and mineral soil depend in part on the amount of fuel consumed. Fuel consumption depends on several factors, but moisture content has the strongest influence (Sandberg and Ottmar 1983). Other key factors that determine consumption in a specific fuel type are preburn amounts, mineral soil incorporation, degree of consumption of adjacent fuels, weather conditions, and ignition technique. Because the largest quantity of fuel (organic matter) at most northern Rocky Mountain sites occurs as duff and large woody material (Brown and See 1981), the influence of fire on these fuel groups is of great interest. These fuel types are potential sources of soil heating and provide much of the nutrient base that determines long-term site productivity (Harvey and others 1989). Thus, controlling the degree of consumption by prescribed burning is important for controlling soil heating and potential site productivity.

In western larch/Douglas-fir (*Larix occidentalis* Nutt./ *Pseudotsuga menziesii* var. *glauca* [Beissn.] Franco) clearcuts in northwestern Montana, Shearer (1975) found duff reduction by burning was closely related to moisture content. Sandberg (1980) used NFDRS 1,000-hour moisture content as a surrogate for duff moisture to predict duff consumption in Oregon and Washington Douglas-fir shelterwoods. Norum (1976) and Harrington (1987) reported duff consumption in western larch/Douglas-fir and ponderosa pine underburns, respectively, to be related to duff moisture and preburn duff amounts. Consumption of surface woody fuels was also important in Norum's (1976) duff-reduction model.

Large woody fuels are important for maintenance of long-term site productivity (Harvey and others 1987), but their consumption by fire is highly variable. Several factors influence consumption of large woody fuels, but their predictive value has not been developed. Moisture content is a key variable (Sandberg and Ottmar 1983). Degree of rot and grouping of fuel pieces are also influential factors (Albini 1976).

A wide range of fuel consumption from prescribed fires and wildfires can be expected because of the variability in factors that affect consumption. In a series of underburns in western larch/Douglas-fir in western Montana, large woody fuel reduction ranged from 0 to 41 tons/acre; small woody fuel reduction ranged from 0 to 6 tons/acre, and duff reduction ranged from 23 to 73 percent of the preburn depths, which averaged from 1.7 to 4.3 inches (Norum 1976). In several clearcut burns in northwestern Montana, duff depth reduction varied from 25 to 93 percent of the 2.5-inch preburn average (Shearer 1975). In northern Idaho, a series of slash underburns in ponderosa pine shelterwood cuts resulted in 0 to 17 tons/acre reduction of large woody fuels and 0 to 56 percent reduction of

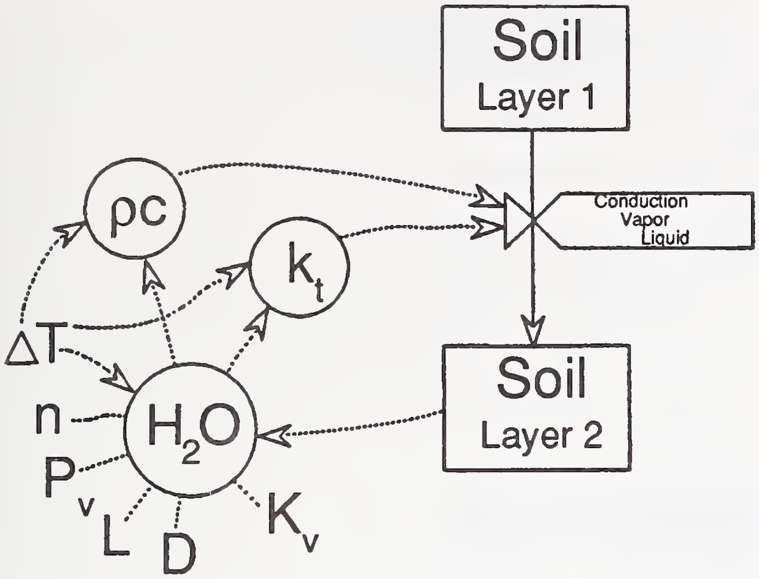

Figure 7—Details of the factors that control heat transfer between soil layers. The circles represent the primary variables (H_2O = moisture content, k_t = thermal conductivity, ρc = volumetric heat capacity) that influence the processes of heat transfer (conduction, vapor movement, and liquid movement) between the layers. T = the temperature gradient, n = the vapor flow enhancement factor, P_v = the vapor density, L = the latent heat of vaporization, D = vapor diffusivity, and K_v = hydraulic conductivity; these are secondary variables that determine state of the primary variables.

the duff, which averaged less than 3 inches before burning (Brown and others 1985). Different amounts of fuel consumption are expected to have different short- and long-term impacts on forested sites.

EFFECTS ON FUEL NUTRIENTS

Combustion of organic matter is a rapid decomposition process in which oxidation changes the form of carbon (C), nitrogen (N), and other mineral elements (Harvey and others 1989). Most of the N found in forest systems is in an organic form that is unavailable for plant use. Physical and microbial decomposition are the general processes by which organic N is transformed into mineral N, but these processes are slow. The products of both biological and combustion oxidation can be similar. In both cases, the result is a mineralization of most nutrients. However, burning results in volatilization of N, phosphorus (P), and sulphur (S) (Wright and Bailey 1982). The degree of volatilization and subsequent loss of nutrients depend on the temperatures produced during burning (DeBano and others 1979) and the amount of duff consumed (Little and Ohman 1988) (fig. 8).

Some of the N volatilized from the duff, however, is mineralized and condensed in the soil (Jurgensen and others 1981; Stark 1977). Direct losses of N from burning ponderosa pine ranged from 70 lb/acre in moderate ponderosa pine underburns (Covington and Sackett 1984) to 800 lb/acre in a severe wildfire (Grier 1975). Other elemental nutrients can be increased in the residual organic matter or soil (Viro 1974), can remain relatively unchanged (Nissley and others 1980), or can be reduced (Feller 1988) depending on consumption and soil heating.

EFFECTS ON SOIL PROPERTIES

Fire effects on the physical and chemical properties of soil depend on the amount of material consumed during burning, the magnitude and duration of soil heating, the frequency of fire, and the postfire environment. High soil heating causes nutrient loss from the soil. Redistribution or losses of organics and nutrients within the soil often change the physical properties of the soil (Wells and others 1979).

Soil Structure and Water Absorption—In general, most fires do not cause enough soil heating to produce direct changes in soil structure. But if temperatures of 400 to 800 °C occur the clay fraction in the soil is altered, making the texture more coarse and erodible (Chandler and others 1983). In some cases soil tilth was improved under the high temperatures of slash burning, because heating made clay more friable (Wells and others 1979). When duff is completely consumed, mineral soil is exposed to higher postfire temperatures and raindrop action, which often reduces soil porosity (Wells and others 1979). DeByle (1981) reported that burning increased bulk density of the soil on burns that varied widely in intensity and degree of soil heating.

The ability of soils to absorb water after fire is directly related to the degree of soil heating and amount of organic matter consumed. Infiltration rate is most often reduced by fire on forest and range soils, but some studies showed increased infiltration rates or no change (Wells and others 1979). Neal and others (1965) suggested that as soil heating increases, soil organic matter decreases, and moisture-holding capacity is reduced. A water-repellent layer is sometimes created when volatiles from burning organic materials are driven downward and condense on soil particles (DeBano and others 1979). Coarse-textured soils became more water repellent than fine-textured soils because the surface area per unit volume of coarse grains is less than that for fine grains. Water movement and water holding capacity are reduced at and below the water-repellent layer. Formation of these water-repellent layers is an important concern on shrublands in California, Arizona, and Oregon (Chandler and others 1983), but is of less importance in the northern coniferous forests (DeByle 1981).

Soil Nutrients—Fire can alter the amount of organic matter and mineral nutrients in the soil just as it does in the duff and slash layers. Organic matter in the soil is destructively distilled between 200 and 300 °C, is charred between 300 and 400 °C, and is consumed above 450 °C (fig. 8). Dyrness and Youngberg (1957) found a 60 percent reduction in organic matter in the upper 2 inches of soil on a portion of a clearcut burn in Oregon. DeByle (1976) reported a 12 percent loss of organics in the surface soils 1 year following several clearcut burns in western Montana, but preburn levels were reached by the second year. Within a clearcut burn in northern Idaho, Niehoff (1985) found the organic matter content in the surface 3 inches

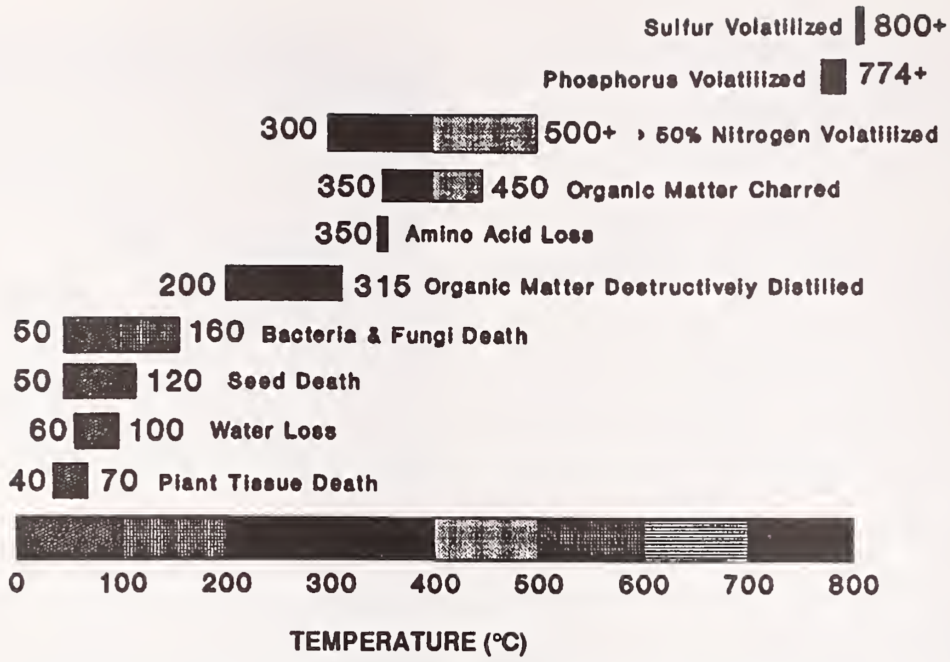

Figure 8—Temperature range for important temperature-sensitive events.

of mineral soil to be only 0.6 percent on extremely burned microsites compared to 3.8 percent in unburned soils. Only 7 percent of the area, however, was classified as extremely burned.

A redistribution of organic matter from the forest floor or slash layers into the mineral soil may occur with light to moderate burning (Wells and others 1979). Increases in soil organic matter were reported for red pine (*Pinus resinosa* Ait.) underburning (Alban 1977) and Douglas-fir clearcut burning (Neal and others 1965). Other soil properties are affected by soil organic matter changes. As soil organic matter increases, total N increases along with other minerals (Lunt 1951; Wells and others 1979). Cation exchange capacity can also increase (Alban 1977). High soil heating may, however, reduce both of these properties (Wright and Bailey 1982). In light to moderate fuel-reduction burns, total N is frequently unchanged for ponderosa pine underburns (Kovacic and others 1986) and for Douglas-fir/western larch clearcut burns (Jurgensen and others 1981). Where soil temperatures reach 300 °C and above (fig. 8), from 50 to 100 percent of the N will be volatilized (White and others 1973).

Cations are generally increased in the soil following their oxidation from burning the surface organic matter (Wells and others 1979). Potassium (K), calcium (Ca), magnesium (Mg), and phosphorus (P) were either increased or unchanged by underburns in ponderosa pine (Harrington 1977), red pine (Alban 1977), and Douglas-fir/western larch (Stark 1977). Losses of these minerals are possible only with high soil heating because volatilization temperatures are greater than 750 °C (Wright and Bailey 1982). With the addition of cations (K, Ca, Mg), commonly acidic forest soils frequently experience an increase in pH (Wells and others 1979). This serves to further increase nutrient availability as well as improve the microenvironment for biological mineralization.

Nitrogen is a key element for plant growth and sites may experience reduced productivity when N is tied up in accumulated duff and fuels. Generally, fire causes an immediate increase in ammonium ions (NH_4+), a readily available form of N (Jurgensen and others 1981). Ammonium levels were elevated for 6 months (Neal and others 1965) to 1 year after fire (Harrington 1977). With the addition of readily available nutrients and an increase in pH, as well as improved moisture and temperature conditions (Bisset and Parkinson 1980), biological mineralization and nitrification can increase on burned sites as long as carbon products are available (Wells and others 1979). These same microenvironment changes can also be favorable for symbiotic (Wright and Bailey 1982) and nonsymbiotic (Harvey and others 1979) N-fixation. Normally, the postfire flush of available nutrients returns to preburn levels with biological and physical immobilization and as the C:N ratio increases. Covington and Sackett (1986) reported that repeated interval underburning in ponderosa pine resulted in additional pulses of available N. If burns consume all of the forest floor and much of the soil organic matter, it follows that mineralization potential can be greatly reduced (Niehoff 1985). Because many stands are N limited, reduced mineralization can limit productivity (Harvey and others 1987).

Erosion—Forest organic matter, both living and dead, is important in controlling soil erosion. Excessive removal of vegetation and forest floor horizons increases erosion potential. Actual soil movement depends on storm intensity, slope steepness, and soil type (Wright and Bailey 1982), but the surface, nutrient-laden soils are the first to be lost. Under extremely dry conditions where most of the forest floor and deep-rooted vegetation is removed by fire, erosion potential is greatly increased. Campbell and others (1977) estimated that a severe storm following a ponderosa pine wildfire removed 13.8 tons of soil per acre. Within 1 year, returning ground cover significantly reduced erosion. Following clearcutting and prescribed burning in a Douglas-fir/western larch cover type, erosion from snowmelt and summer storms was temporarily increased (Packer and Williams 1976). With the exception of the most intense burns on south slopes, soil movement was reduced within 3 years. Vegetation recovery and organic matter deposition on bare soils are important for reducing erosion potential.

Slower vegetation recovery is associated with greater soil heating (Lyon 1966; Morgan and Neuenschwander 1988), which affects the length of time the soil is susceptible to erosion.

EFFECTS ON VEGETATION

Plant adaptations to climate and disturbance, including fire, were grouped and classified by several investigators (Grime 1979; Lyon and Stickney 1976; Noble and Slatyer 1977; Raunkiaer 1934; Rowe 1983). These classifications are working hypotheses of how species adaptations function, and were used as a basis for some plant succession models (Cattelino and others 1979; Hungerford 1987; Keane 1987; Morgan and Neuenschwander 1988) and for describing plant response to fire (Fischer 1989; Fischer and Bradley 1987). The relative resistance of plant species to damage from fire depends on the degree and duration of soil heating, the depth of perennating plant parts, heat resistance, and colonization potential (Morgan and Neuenschwander 1988; Noste and Bushey 1987). The following discussion reviews lethal temperature thresholds for plants, factors that influence damage, and species rooting habits.

Lethal Temperature and Heat Thresholds—The concept of stress resistance (Levitt 1980) is useful for understanding plant responses to heat. Stress resistance represents the ability of plants to survive and adjust to stress. A plant may avoid heat by insulating itself from the heat source; for example, the thick bark insulates the cambium from fire. Some plants tolerate heat by coming to thermal equilibrium with the heat source without suffering injury. Some species evade heat. For example, a plant species that seeds in from unburned areas will have evaded heat stress by escaping the stress.

Numerous workers have identified lethal temperatures for living plant tissue. Hare (1961) and Levitt (1980) published extensive literature reviews that summarized plant response to temperature. Levitt (1980) summarized the lethal high temperature thresholds for different plants and plant parts. Lethal temperatures range from 42 to 70 °C for vascular plants, and from 24 to 110 °C for nonvascular plants. Conifer seedlings are generally killed by temperatures between 50 and 60 °C (Baker 1929; Hare 1961); leaves may be killed at temperatures from 42 to 55 °C. Seeds are generally more resistant to heat, with lethal temperatures ranging from 60 to 120 °C. Limited studies on heat tolerance of roots (Hare 1961) show that lethal temperatures range from 48 to 54 °C.

Factors Influencing Plant Response—A variety of factors influence the lethal temperature of plants and plant parts. Duration of exposure to a given temperature is recognized by most investigators as very important for determining the lethal temperature. Temperature magnitude and duration are controlled by the heat source and the medium of transfer. The lethal temperature of plant tissue varies inversely with exposure time (Hare 1961; Levitt 1980), and the relationship is exponential (Levitt 1980; Wright and Bailey 1982). Baker (1929) observed that heating tissues of conifer seedlings to temperatures of 50 °C for long periods did not cause damage, but temperatures a few degrees higher (54 °C) caused death within a few minutes. Differential species responses also exist (Baker 1929; Flinn and Pringle 1983).

The metabolic state of the plant and the moisture content of the tissues are also important. Dormant tissues generally tolerate higher temperatures better than actively growing tissues (Hare 1961; Levitt 1980). Some seeds, for example, are able to survive temperatures of 120 °C in dry conditions but only 100 °C in boiling water, if they have not imbibed water. Lethal temperatures decrease for seeds with higher moisture contents. In maize seeds, the lethal temperature range was 45 to 50 °C at moisture contents greater than 40 percent, but was 80 °C at moisture contents of less than 10 percent (Levitt 1980). The relationship between moisture and lethal temperature also holds for actively growing tissues (Levitt 1980). These and other data led some investigators to postulate that the moisture content of underground plant parts influences lethal temperature, and further, that dry heat and wet heat may differentially influence lethal temperature (Sapper 1935). The moisture/lethal temperature relationship may be partially explained, because moisture alters the heat-transfer properties and physiological activity of the tissues.

Plant age, vigor, and season have been related to fire response (Noste and Bushey 1987), but it is not clear how responses are related to prefire environmental stress or whether heat resistance is altered. Season of burning influences plant survival and resprouting ability (Flinn and Pringle 1983; Flinn and Wein 1988; Hare 1961; Noste and Bushey 1987). Harrington (1989) postulated that the seasonal effect was related to carbohydrate levels in roots. Fire and other disturbances are most detrimental when carbohydrate reserves in the roots are at the minimum. This is probably directly related to reduction of photosynthetic capacity and the cost of its replacement rather than direct heat damage to roots or sprouting parts. Julander (1945) showed that any treatment favoring accumulation of root reserves increased the growth response of five grass species. Dormant buds are possibly more heat resistant than active buds, thus less susceptible to damage from the downward heat pulse of a fire. Flinn and Pringle (1983) tested the heat tolerance of rhizomes in different seasons, and found that season had a direct effect on the number of sprouts for several species. This response was assumed to be related to stored nutrient reserves.

The size of rhizomes, root crowns, roots, and other structures should influence the amount of heat needed to raise the temperature to lethal levels, thus avoiding the detrimental effects of heat. Flinn and Pringle (1983) studied heat effects on rhizomes, but did not find any relationship between the surface-to-volume ratio and avoidance of high temperatures. Their experimental methods may have masked species differences and effects of surface-to-volume ratio. Larger structures should buffer temperature changes within their tissue. Depending on thermal properties, internal temperatures of these structures may be considerably lower than temperature of the surrounding soil especially when heat pulses are of short duration.

Species Rooting Habits and Location—Bradley (1984) reviewed a variety of approaches for classifying fire adaptations. Although some of these classifications

are not related to fire response the concepts are useful. Raunkiaer's (1934) life-form system and subsequent adaptations (Gill 1981; Keeley 1981; Stickney 1986) accounted for the position of meristematic tissue relative to the ground surface. The life-form system is quite effective for evaluating survival potential, but it does not consider colonization potential, except for residual seed banks within the soil.

In spite of the importance of the morphology and depth of perennating organs, only a small number of studies describe the morphology and location of underground regenerating structures. McLean (1969) described structures for plants in forests of British Columbia; Flinn and Wein (1977) described structures in the Acadian Forest of New Brunswick, and several studies described structures for plants in the northwestern United States (Antos and Zobel 1984; Bradley 1984; Miller 1977).

McLean (1969) identified the rooting characteristics of 26 species; Flinn and Wein (1977) sampled the depth of 34 species, and Bradley (1984) evaluated the depth and morphology of eight species. McLean (1969) identified six types of structures based on rooting habit with reproductive tissues located within the duff layers and in the upper 13 cm of mineral soil. Bradley (1984) found rhizomes at depths from 0 to 65 cm in the mineral soil. Flinn and Wein (1977) reported that reproductive structures were found in the litter and duff layers for 50 percent of the species, and below 4 cm for 18 percent of the species. They also found that the depth of reproductive tissue for a particular species varied from 4 to 14 cm between sites and within 2 cm on a site. Both Bradley (1984) and Flinn and Wein (1977) found that depth of reproductive tissues is species specific.

McLean (1969) identified three categories of fire response that are related to rooting structure and depth. Plants with reproductive structures in the litter and duff were susceptible to fire damage. Structures between 1.5 cm and 5 cm deep were intermediate in response and those at depths below 5 cm were not damaged by fire. Bradley (1984) noted sprouting on burned sites from depths as great as 30 cm; she observed that rhizomes with greater mass seemed to have a greater survival potential and deeper rhizomes were more likely to have viable buds after a fire. Flinn and Wein (1977) hypothesized that the deeper the reproductive tissue, the more likely they are to avoid fire damage. They also hypothesized that variations in species response to fire are largely due to interaction between depth (which is species related) and soil heating differences.

EFFECTS ON MICROORGANISMS

Bacteria and Mycorrhizae—The effects of fire on soil microorganisms are complex and varied. Fire impacts microbial numbers and activity directly by heat and indirectly by physical and chemical changes in the postfire environment. Generally, the greater the consumption of duff and higher soil heating, the greater the immediate negative impact on the microorganisms. Populations found near the soil surface or within the forest floor were more easily killed than those found in deeper horizons (Ahlgren 1974). Light burning, even on a periodic basis, may have minimal effects on most microbes (Jorgensen and Hodges 1970). Fires of long duration, such as those under slash piles and smoldering duff fires, reduced populations at depths to 25 cm below the surface (Renbuss and others 1973). Actual temperature effects on microbes depend on the type of organism and soil moisture. In general, bacteria are more resistant to heating than fungi. Lethal temperatures for bacteria were reported to be 210 °C in dry soils and 110 °C in wet soils (Dunn and DeBano 1977). In forest soils, Ahlgren and Ahlgren (1965) found bacteria numbers were reduced by heating to 200 °C. Nitrifying bacteria are less resistant than others and were mostly destroyed at 140 °C in dry soils and 75 °C in moist soils (Dunn and DeBano 1977). Saprophytic fungi in chaparral soils were killed at temperatures above 120 °C and 60 °C in dry and wet soils, respectively, but "heat shock" fungi prevailed up to 155 °C in dry soils and 100 °C in moist soils.

Following the initial reduction in microorganism populations, a several-fold increase frequently occurs (Jurgensen and others 1981; Neal and others 1965). Increases are attributed to reinfection of the burned sites and an improved environment for microbial activity, which includes more favorable temperatures, pH, soil moisture, and nutrient levels (Ahlgren 1974). Recovery of nitrifying bacteria appears related to levels of soil heating. Wells and others (1979) reported slow recovery of these bacteria following severe fires, but Jurgensen and others (1981) and White (1986a) found increased populations several weeks after light-to-moderate burns. White (1986b) also indicated that fire may release allelopathic chemicals that inhibit nitrification in ponderosa pine forest floors and soils. Nonsymbiotic nitrogen fixing bacteria also responded positively to burning in several forest types (Lunt 1951).

Soil fungi, which are important in organic matter decomposition and nutrient release, respond similarly to bacteria. An increase in heterotrophic fungi is thought to be responsible for additions of nitrate-N following burning (Wells and others 1979). The response of fungi-producing mycorrhizal roots appears to be related to the prefire nutritional status of the organic and mineral horizons, and to the associated plant species. Mycorrhizal fungi often decrease with burning because these associations occur more frequently in the organic horizons, which can be consumed (Harvey and others 1989). Additionally, mycorrhizae may not form on forest tree seedlings following burning because of increased soil fertility (Wright and Tarrant 1958). However, Black (1985) found more mycorrhizal short roots on Douglas-fir seedlings following a "hard" burn that apparently improved the N status of the soil. Therefore, either increases or decreases in mycorrhizae may be found following burning depending on the degree of soil heating and whether or not critical nutrient levels are achieved.

Root Disease Fungi—Fire is effective in controlling some diseases (Ahlgren 1974; Dickman and Cook 1989; Froehlich and others 1978), but fire may also increase disease by providing infection courts (Ahlgren 1974; Hepting 1935; Littke and Gara 1986; Nordin 1958), by heat stimulation of organisms (Ahlgren 1974), or by providing favorable environments (Ahlgren 1974; Munnecke and others 1981). Most effects of fire are indirect rather than direct. For example, Froehlich and others (1978) were unable to identify the mechanisms by which fire controlled *Heterobasidium annosum*; but they suspected microclimatic changes,

since neither soil pH, chemistry, nor moisture were altered. Thus, the effect appeared to be indirect on the pathogen or the inoculum potential.

The most probable effects of fire on *Phellinus weirii* are indirect, through the effect on host species dynamics and on the longevity of inoculum. Fire influenced the area and pattern of infestations of *P. weirii* in a mountain hemlock forest (Dickman and Cook 1989) by reducing the area of infestation, although the pathogen was not eliminated. Fire killed the more susceptible hemlock and led to increased dominance of lodgepole pine, which is fairly resistant to *P. weirii* damage. Although fire may affect reductions in inoculum by killing the pathogen, enough inoculum may remain to reinvade. Infestation centers may not be manifested everywhere, but the possibility exists that the fungus is hidden (not expressed) in areas where trees are apparently healthy (Dickman and Cook 1989). The lack of expression, or seeming lack of presence, may be due to the fire regime of the area. It is also quite likely that *P. weirii*-caused mortality creates fuel, enhancing the probability of fire, which, in turn, limits expression of the disease.

EFFECTS ON POSTFIRE ENVIRONMENT AND STAND DEVELOPMENT

Solar Radiation and Soil Temperature—Overstory removal by harvesting or burning increases solar radiation received by the ground surface (Hungerford 1980). Burning not only removes shade, but changes the surface albedo (Hornbeck 1970; Hungerford 1980; McCaughey 1978), which alters the energy balance at the ground surface. Net radiation is greater on burned surfaces than on exposed slash or mineral soil surfaces (Hungerford 1980). Increased solar radiation results in elevated soil temperatures. Exposed soil surface temperatures averaged as much as 36 °C warmer at midday and 15 °C cooler at night than those on unburned forested sites (Hungerford and Babbitt 1987). These extreme temperatures cause mortality of regenerating tree seedlings (Hare 1961; Shearer 1967). Temperatures in the duff (if it is not consumed) and in the underlying soil are also warmer than those in the undisturbed forest. At a depth of 2 inches, temperatures averaged 7 °C warmer (Hungerford and Babbitt 1987). Character of the surface (burned, litter, slash, or bare mineral soil) also significantly influences temperature of the soil to a depth of 16 inches. The duration of these extreme temperature differences is influenced by vegetation recovery rate, which in turn influences regeneration.

Soil Moisture—The moisture status of soils following burning is greatly influenced by organic matter consumption and soil heating. Exposure of burned mineral soil may reduce infiltration and percolation when ash and charred organic matter fill soil pores (Wells and others 1979). Tarrant (1956b) found that percolation rates decreased following severe burns, but increased following light burns. Reduced moisture interception by the forest floor allowed more moisture to reach the mineral soil (Aldon 1968). Haase (1986) reported that soil moisture increased during the growing season following a moderate ponderosa pine underburn. Vegetation removal results in increased soil temperatures and evaporation rates and decreased foliar interception and transpiration losses (Wells and others 1979). The formation of a hydrophobic layer by burning can lead either to a reduction in soil moisture, if the layer is near the soil surface, or to an increase in surface soil moisture if the repellant layer reduces percolation in subsurface horizons.

Plant Regeneration—The reestablishment of vascular plants on burned sites depends on the type, quantity, and morphology of the species present before burning, the characteristics of the fire, and the postfire environment (Morgan and Neuenschwander 1988; Rowe 1983; Stickney 1986). A low-intensity fire that consumes only surface litter fuels generally has little impact on most existing vegetation and results in minimal and short-lived changes in the microenvironment. As fire intensity, fuel consumption, and soil heating increase, resulting in greater damage to vegetation, the contrast in environments from preburn to postburn will also increase.

Natural regeneration of conifers in the northern Rocky Mountains is entirely by seed; therefore, seedbed characteristics are vital to seedling establishment. Seedlings of different conifer species have different environmental requirements for establishment. Generally, seral species establish and grow better than competing climax species in warmer, open environments that result from a fire. Western larch and both seral and climax ponderosa pine regenerate more readily on mineral or lightly duff-covered soils than they do on thick duff (Haase 1986). Thick duff generally dries more rapidly from the surface downward than mineral soil, leaving insufficient moisture for initial seedling root establishment (Boyce 1985). Extreme moisture deficits commonly occur in duff between rains (Potts 1985).

Root productivity is key to survival and growth of plant seedlings. A large root system increases survival potential on drier sites and promotes greater aboveground biomass production on both wet and dry sites by tapping a larger reservoir of nutrients and moisture. Postfire soils with reduced organic horizons influence root activity in several ways. Because root growth is affected by soil temperature, a warmer postburn soil may promote greater root expansion (Hermann 1979). Increased moisture levels in the mineral soil resulting from duff reduction should also favor root growth (Hermann 1979). However, warm and dry conditions on south exposures could be detrimental to root growth and cause root mortality.

Increased soil nutrient concentrations following fire can have various impacts on root response, depending on the initial nutrient status. If nutrients are limiting growth prior to burning, an increase following burning would likely improve root production. If preburn nutrients are less limiting, a large pulse of available nutrients following fire could lead to greater shoot-to-root ratios (Hermann 1979). However, Harrington (1977) found ponderosa pine seedlings growing on burned seedbeds had significantly larger shoots and roots than those on unburned seedbeds. A positive relationship between duff consumption, nutrient levels (especially ammonium-N), and seedling size was implied. Eramian and Neuenschwander (1989) reported greater leader growth of Douglas-fir seedlings on burned sites than on bulldozer-prepared sites, due to warmer soils

and more available moisture on the burned sites. Graham and others (1989) emphasized that the degree of burn represented by the amount of duff consumed was important in the evaluation of site preparation for prescribed burning for artificial regeneration. In warm, moist habitat types, burning the surface litter and fine slash fuels, but not the duff layer, was deemed important to optimize growth of planted seedlings.

Reduction of competition is also important for conifer seedling survival and growth. Fire intensity will determine, to a great extent, the type and amount of postfire shrubby and herbaceous competition. Sites with greater soil heating usually are dominated by offsite colonizers; Sprouters are common following more moderate fires (Fischer and Bradley 1987; Ryan and Noste 1985). Sprouters are generally more competitive because they are already established. Walstead and Seidel (1990) reported that Douglas-fir seedlings in broadcast-burn clearcuts were larger and had better survival than those in unburned clearcuts. These differences were attributed to reduced competition for the first 3 years of seedling establishment.

SUMMARY AND CONCLUSIONS

Presettlement fire was an important cycling process in forests of the Interior West. Soil properties and plant succession were kept in a dynamic state through fuel consumption and soil heating. Recent forest management activities, primarily fire suppression and harvesting seral species, have led to major changes in many forest types. Organic matter accumulation is greater than in presettlement times, and dense stagnant stands of climax species are common. Because fuel consumption and environmental factors determine the degree of soil heating, which in turn governs the extent of the impact on soil properties, managers must be aware of potential fire effects. Fires that create light to moderate soil heating facilitate nutrient cycling, making nutrients more available to plants and microbes, but soil productivity and stability are adversely affected by excessive heating.

Soil heating (depth and duration of high temperatures) varies widely between fires and within a particular fire because of differences in fuel characteristics, weather conditions, and soil properties (particularly moisture content). Many effects on soil physical, nutrient, and biological properties are related to the amount of organic matter consumed and temperature levels. The physical properties of mineral soil can be changed by heat altering the internal soil structure and by changing the amount and form of carbon compounds. Nutrients can be volatilized, transformed into highly available ions, or remain unchanged by burning. These impacts are directly related to fire-caused temperatures. Temperature ranges and durations also influence survival of microorganisms, plant regenerating parts, and roots. Reestablishment of both macro- and microflora and fauna is dependent on the postfire environment, which, again, is partially determined by the fire effects.

Burning almost always results in redistribution or loss of nutrients (especially N), killing of some plants, changes in postfire environment, and alteration in microbial populations. These changes can be minimized by avoiding fires that cause high soil heating. Prescriptions for sites having shallow, nutrient-poor soils need to be carefully designed and carried out so that nutrient loss and erosion potential are minimized (Grier and others 1989; Harvey and others 1989). Reducing large fuel accumulations may cause high soil heating, which may be needed to manipulate plant species to meet a vegetation management objective, or effectively manage a disease problem. Therefore, the objectives should be very clear and the prescriptions should be well designed to provide only the level of soil heating required. Recent research information should help in the development of fire prescriptions that distribute the heat load (Brown and others 1985; Frandsen and Ryan 1986), allow us to predict the level of soil heating (Campbell and others 1990; Hungerford 1990), and assist in the prediction of nutrient changes associated with the predicted soil heating level (Harvey and others these proceedings; Jurgensen these proceedings). Prescribed fire for site preparation and fuel reduction should cause no more disturbance than is actually necessary to accomplish the objectives and maintain or enhance long-term site productivity. Much more research is needed to better define measures of productivity, to apply systems-analysis approaches for solving the complex relations between the factors that influence productivity, and to elucidate the effects of management practices on long-term productivity (Grier and others 1989).

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Speakers answered questions from the audience after their presentations. Following are the questions and answers on this topic:

Q.—Does fire and fire recurrence interval influence the severity of root disease in conifers?

A.—A number of associations between fire and root disease can be found in the literature, but causal mechanisms are poorly understood. Fire, or its exclusion, can significantly affect the expression of root disease through its impact on stand structure and species composition. Recurring fire favors seral conifer species that are less susceptible to infection, damage, and mortality from root disease pathogens. The presence of seral species in a healthy stand structure may retard pathogen development and mask root disease expression. Root diseases are generally more prevalent in stands where fire has been excluded for decades. The physical and chemical microenvironment is also altered by fire, which can affect population dynamics through innoculum potential and interaction with competitive organisms. Heat generated by fire can be sufficient to kill pathogens in soils and unburned organic matter, but it is not clear what the long-term effects on disease are.

Q.—Are there management activities that have effects similar to hot fires? How fast or slow is recovery?

A.—No. Management activities other than fire, such as mechanical removal or chemical treatment, may have similar apparent results, but there are considerable qualitative differences. The same amounts of organic material may be treated or removed, which may similarly reduce fire hazard, but great differences in soil chemistry, physical properties, and productivity may occur. No activity other than fire heats the fuel and soil and alters the physical and chemical properties in the same manner. The rate of recovery will depend on the type and intensity of management activity, and is dependent on the specific part of the soil system one is interested in. For example, population levels of some microorganisms may recover and even exceed pretreatment levels within a few weeks of treatment, but nutrients may take years to return to pretreatment levels. More productive sites tend to recover faster than cold, dry, and harsh sites.

Q.—Are site preparation/slash burns different than the periodic wildfires historically? Do these two types of fires differ a great deal?

A.—Prescribed fire effects can be similar or different than those of historical wildfires. Differences between the two fire types are primarily related to differences in fuel consumption and soil heating, which are influenced by fuel, weather, and soil conditions. On warm dry habitats, historically frequent surface fires maintained light fuel loads. These fires caused minimal soil heating. Under current management, fuel loads on warm, dry habitats are greater after harvesting than commonly occurred historically. Therefore, even though prescribed burns are conducted during more moderate conditions, more fuel is generally consumed, which causes greater heat impacts than occurred during presettlement fires. In more moist habitats, large amounts of fuel accumulated between infrequent historic wildfires. These fires likely caused considerable soil heating. Under current management, fuel loads on moist habitats are similar to presettlement times, but prescribed burns are not conducted during the extremely dry conditions when historic wildfires occurred. Therefore, soil heating and impacts on soils for prescribed fires are likely to be less than for historic wildfires because of differences in fuel, weather, and soil conditions.

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RHIZOSPHERE BIOLOGY: ECOLOGICAL LINKAGES BETWEEN SOIL PROCESSES, PLANT GROWTH, AND COMMUNITY DYNAMICS

Randy Molina
Michael Amaranthus

ABSTRACT

Productivity of the forest plant community results from interactions of shoots and roots with the environment. One of the more important and least understood biotic zones of interaction is the soil immediately surrounding and influenced by roots, known as the rhizosphere. Microorganisms and processes in the rhizosphere profoundly influence plant growth through effects on uptake, storage, and cycling of nutrients, suppression of pathogens, and development of soil structure. Rhizosphere organisms are affected by and contribute to the successional dynamics of forest communities. Protecting the functional biodiversity of rhizosphere organisms through proper management practices is essential to maintaining ecosystem health and resiliency.

INTRODUCTION

Today's forest management decisions integrate increasingly complex and changing social, economic, and ecological values. Foresters not only are concerned with the basic silvicultural goals of harvesting and regeneration but also seek to sustain long-term productivity of forest sites, protect nontimber resources such as water and scenery, and maintain biodiversity. All this requires understanding of how forest ecosystems function. Today, forestry and forest sciences are shifting dramatically from basing decisions on how trees operate to a holistic integration of how all the parts of a forest work together, how they are linked and interdependent.

The study of soil biology also has shifted fundamentally. Over the last decade, research has shown the critical linkage of soil microorganisms to forest productivity and community dynamics. Ecosystem studies quantified the tremendous energy invested by trees in fine root production and organisms growing in the immediate vicinity of these roots, known as the rhizosphere. As much as 80 percent of the photosynthate of trees is used to support fine roots and associated microorganisms (Fogel and Hunt 1983; Vogt and others 1982). Laboratory and field studies have shown that roots of the same or dissimilar plants can be connected by commonly shared symbiotic root fungi, the mycorrhizal fungi; carbon (sugars) and other nutrients can actually flow between connected plants (Read and others 1985). Increasing attention has been focused on the food web dynamics of soil organisms, particularly their relations to the vitality of highly visible aboveground organisms (Ingham and others 1985). An increasingly sophisticated and knowledgeable public is becoming aware of these tight ecosystem linkages. Such heightened awareness and demand for sustaining total ecosystems requires that forest managers stay current on the latest developments in the many disciplines of forest research.

Our previous research has concentrated on mycorrhizal fungi of forest trees, with emphasis on selection of beneficial fungus strains for inoculation of nursery seedlings. We tried to enhance seedling survival and growth on hard-to-regenerate sites by tailoring root systems with fungi adapted to the planting site. In addition to successfully developing this technology (see Castellano and Molina 1989), we encountered ecological concepts relevant to managing forest sites and protecting soil organisms. We found, for example, that the tremendous biological diversity of mycorrhizal fungi represents a similarly wide range of physiological diversity providing a wide range of benefits to plants in diverse habitats. Different site disturbances and patterns of reforestation affect the survival potential of mycorrhizal fungus propagules in the soil. Enhancement of seedling performance by mycorrhizal inoculation was most successful on drought-stressed sites with low population levels of residual mycorrhizal fungi. Such findings allowed development of testable hypotheses on how forest practices degrade the soil biological community in ways requiring inoculation. Our major goal now is to achieve better understanding of the natural forest processes that maintain viable populations of soil microoganisms after natural disturbance and thus contribute to forest recovery; that is, natural soil biological mechanisms of resiliency (see Perry and others 1987).

The objectives of this paper are to (1) overview fundamental concepts in forest soil biology, including key organisms and processes; (2) discuss rhizosphere ecology in the context of interactions and interdependencies of aboveground and belowground biotic communities; (3) emphasize protection of the soil biological community through wise resource management; and (4) update forest managers on research directions in forest soil biology.

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THE RHIZOSPHERE OF FOREST PLANTS

Productivity of plant communities results from interactions of shoots and roots with the environment. One of the more important and least understood biotic zones of interaction is the soil immediately surrounding and influenced by roots—the "rhizosphere." Depending on site quality, forests can divert 50 to 80 percent or more of the net carbon fixed by photosynthesis to belowground processes (Fogel and Hunt 1983; Vogt and others 1982). Although much goes to root growth, a relatively high proportion may be used to feed soil microorganisms and fuel processes in the rhizosphere. This is not energy lost from the plant. These organisms and processes in the rhizosphere influence plant growth positively by enhancing nutrient uptake, storage, and cycling, moisture storage, pathogen suppression, development of soil structure, and protection against environmental extremes. The specialized rhizosphere microorganisms include mycorrhizal fungi, saprophytes, microfauna, pathogens, and growth-promoting and deleterious bacteria (see Curl and Truelove 1986; Linderman 1986 for more detail).

Factors that influence rhizosphere organisms include the age and kinds of plant, soil physical conditions, temperature, moisture, interactions with other soil microbes, and cultural practices (Katznelson 1965). Rhizosphere activities fluctuate in response to changes in plant, microbial community, edaphic, and environmental factors.

Unfortunately, except for mycorrhizal fungi of forest trees and nitrogen fixation in forests, most of our knowledge of rhizosphere dynamics comes from agricultural soils and crop research. Forest soils and processes differ fundamentally from agricultural soils; we are just beginning to probe the complex interactions within the rhizospheres of forest plants and their impacts on forest health and productivity. The following sections discuss key rhizosphere organisms and processes and their importance at forest community and ecosystem levels.

RHIZOSPHERE VERSUS BULK SOIL

Biological activity and function is not distributed homogeneously across the soil environment. Intense biological activity typically extends about 2 mm out from fine roots and differs significantly from that in the bulk soil in numbers, types, and functions of organisms. Fine roots maneuvering through the soil exude amino acids, carbohydrates, and other compounds that stimulate growth of bacteria, actinomycetes, and fungi. These, in turn, produce compounds that repel or stimulate other soil organisms. The microflora is a prime food for "grazer" herbivores such as mites, nematodes, and springtails, which, in turn, fall prey to carnivores such as centipedes and spiders. Saprophytes decay the dead remains of microbes and roots in the rhizosphere and decompose complex organic molecules into basic components. Nutrients and waste released through decomposition are captured and returned to the host plant via mycorrhizal fungi. The fine roots of forest plants and their associated mycorrhizal fungi form an often contiguous network in the upper soil profile. This rooting zone is essentially a rhizosphere zone in which the vast majority of soil nutrients are recycled and retained.

Rhizosphere organisms receive nutrients primarily from root exudates, and nonrhizosphere organisms use organic residues in varying stages of decomposition (Rambelli 1973). The different microbial compositions of bulk and rhizosphere soils reflect these nutritional modes. The rhizosphere microenvironment also differs strongly from that of the bulk soil. Organic substrates and compounds of all kinds are more abundant in the rhizosphere than in bulk soil. This includes humic compounds, polysaccharides, hormones, chelators, and enzymes (Perry and others 1987). The supply of other nutrients such as N, P, Mn, and Fe may limit plant growth because of intense utilization and competition (Foster 1988).

Rhizosphere organisms experience large fluctuations in pH as different ions are removed from the soil by roots and replaced by balancing ions. Similarly, rhizosphere organisms may experience very low water potentials as water is removed by high transpiration rates. High osmotic potential can also occur from the accumulation of calcium at the root surfaces. Local microaerophyllic or anaerobic conditions can develop in the rhizosphere due to high respiration rates by roots and microorganisms (Foster and Bowen 1982), further influencing microbial composition and processes. Rhizosphere organisms can also be buffered from unfavorable environmental changes by gels and polysaccharides secreted by roots and microorganisms.

MYCORRHIZAE AND THE MYCORRHIZOSPHERE

Mycorrhiza literally means "fungus root" and represents the intimate association between the fine roots of plants and specialized symbiotic fungi. Readers are referred to recent reviews of the tremendous body of literature on this mutualistic symbiosis for greater detail (Castellano and Molina 1989; Harley and Smith 1983; Perry and others 1987). We emphasize mycorrhizae in this section and others because most land plants depend on mycorrhizae for nutrient uptake, and mycorrhizae perform far-reaching ecosystem functions in our forests.

Mycorrhizae come in several types and involve thousands of fungus species. Understanding the overall variety of mycorrhizal types, differences between groups of plants in their mycorrhizal association, and physiological and ecological differences among the fungi is key to understanding their biotic diversity and impact on ecosystem function and community development. Ectomycorrhizae, vesicular-arbuscular mycorrhizae (VAM), and ericaceous mycorrhizae prevail in temperate forests.

Ectomycorrhizae predominate in temperate forests of the Pacific Northwest because species of the Pinaceae, Betulaceae, and Fagaceae all form this type. Ectomycorrhizal fungi form an often colorful sheath or mantle around feeder roots and penetrate the root cortex to form the "Hartig net," a zone of nutrient exchange between fungus and host (see figs. 1-5). These fungi also extensively colonize the soil, often to form visible, persistent mats in the upper soil and humus. Most fleshy mushrooms that fruit on the forest floor as well as the hidden subterranean truffles are fruiting bodies of the ectomycorrhizal fungus colonies in the soil. A walk through the woods during the mushroom season reveals the abundance and diversity of

these fungi; thousands of ectomycorrhizal fungus species associate with trees in the Pacific Northwest.

The VAM are also widespread in our forests but have received less attention because they primarily occur on understory species and herbaceous plants. They are typical on cedars (*Thuja*, *Chamaecyparis*, and *Libocedrus*) and redwoods (*Sequoia* and *Sequoiadendron*), however. Unlike the ectomycorrhizal fungi, VAM fungi do not strikingly modify root morphology; they ramify in fine roots to form specialized structures (arbuscules and vesicles). Although they do not form the showy mycelia of the ectomycorrhizal fungi in the soil, they, too, extensively colonize the soil and bring in nutrients to the plants. The VAM fungi only number in the hundreds, and most can form mycorrhizae with a wide range of plant species.

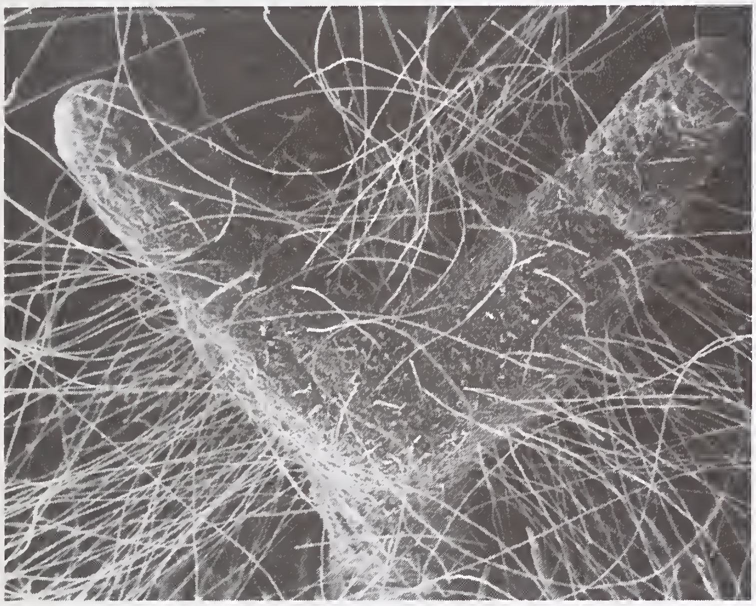

Figure 1—Scanning electron micrograph of pine ectomycorrhiza showing the typically forked branching pattern. Note the numerous hyphae radiating out from the fungus mantle. (Magnification = 93x.) (Photo courtesy of Drs. H. B. Massicotte and R. L. Peterson, University of Guelph, Ontario, Canada.)

Ericaceous mycorrhizae, as the name implies, are formed exclusively with members of the Ericales; for example, *Gaultheria*, *Rhododendron*, and *Vaccinium*. Although this seems a narrow grouping, the Ericaceae are widespread understory components in coastal and mountain forests. Similarly to VAM, the ericaceous mycorrhizal fungi ramify through the fine, hairlike rootlets, to form dense hyphal coils but not to modify the root structure as do ectomycorrhizal fungi. Relatively few fungus species form this type of mycorrhizae, but they seem to be widespread in the soil.

Benefits of mycorrhizae to plants include enhanced uptake of nutrients and water, protection against pathogens, improved resistance to drought, enlarged root systems, and tolerance of heavy metal. The uptake of immobile ions such as phosphorus, typically benefiting growth of hosts, is mostly due to the ability of the fungus hyphae to explore a soil volume for nutrients far beyond the capabilities of the roots. Ectomycorrhizal fungi also produce hormones that promote branching of feeder roots, increasing not only the absorbing root surface but also the contact and exchange zone between fungus and plant. A recent study (Read 1987) also shows an important role for nitrogen uptake by mycorrhizal fungi, particularly from organic nitrogen sources. For example, ericaceous mycorrhizal fungi can degrade protein as well as other organic nitrogen sources (Read 1987). Such an ability may be critically important in organic substrates such as rotting logs or buried wood, common niches for ericaceous plants.

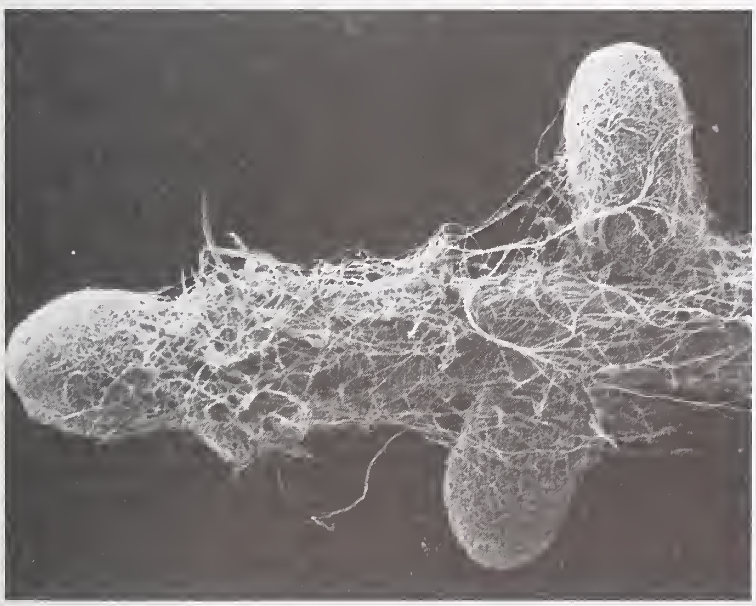

Figure 2—Scanning electron micrograph of a eucalypt ectomycorrhiza showing a pinnate branching pattern and a well-developed mantle. (Magnification = 77x.) (Photo courtesy of Drs. H. B. Massicotte and R. L. Peterson, University of Guelph, Ontario, Canada.)

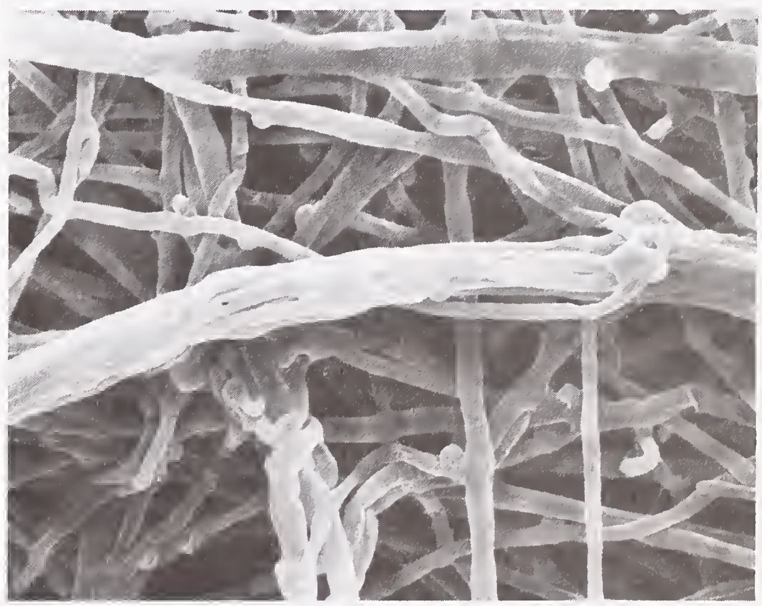

Figure 3—Scanning electron micrograph of an ectomycorrhizal mantle showing the complex development of interwoven hyphal strands and individual hyphae. (Magnification = 1310x.) (Photo courtesy of Drs. H. B. Massicotte and R. L. Peterson, University of Guelph, Ontario, Canada.)

The plants pay for such benefits in photosynthate shuttled to roots. Indeed, mycorrhizal fungi are strongly dependent on a continuous supply of plant sugars and other organic compounds such as vitamins. The fungus functions as a structure of the root system supported by plant energy. The mycorrhizal fungi produce organic exudates and undergo rapid turnover, thereby attracting other microorganisms that feed in the hyphal zone. In essence then, mycorrhizae extend rhizosphere to a zone termed the "mycorrhizosphere" (Rambelli 1973).

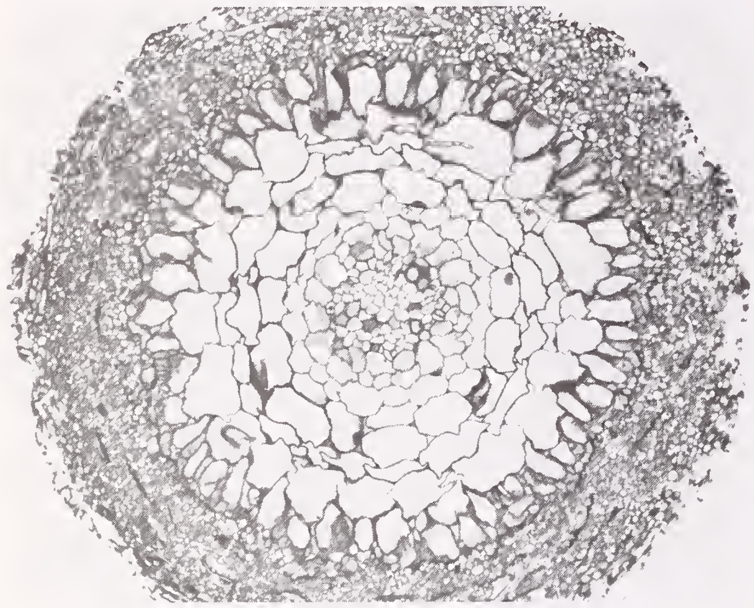

Figure 4—Transverse section through an ectomycorrhiza of red alder. Note the well-developed fungal sheath or mantle. (Magnification = 154x.) (Photo courtesy of Drs. H. B. Massicotte and R. L. Peterson, University of Guelph, Ontario, Canada.)

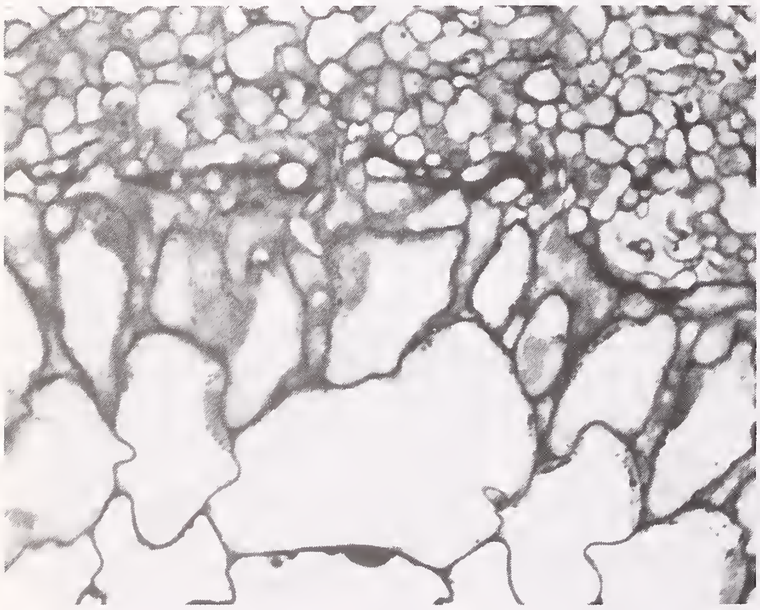

Figure 5—High-magnification transverse section of red alder ectomycorrhiza showing fungal penetration into the epidermis to form the Hartig net. (Magnification = 655x.) (Photo courtesy of Drs. H. B. Massicotte and R. L. Peterson, University of Guelph, Ontario, Canada.)

Most plant benefits noted here were discovered in comparisons of mycorrhizal with nonmycorrhizal seedlings. We are only beginning to understand the role of mycorrhizal fungi and other rhizosphere organisms in forest community development and ecosystem function.

NITROGEN FIXATION

Nitrogen is typically the most limiting nutrient in Pacific Northwest forests. Natural inputs of nitrogen through the process of nitrogen gas fixation are essential to maintain long-term forest productivity in much of the Western United States.

Symbiotic nitrogen fixation can add substantial nitrogen continuously to Pacific Northwest forests (Wollum and Youngberg 1964). Common nodulated plants such as lupine, alder, and snowbrush (*Lupinus*, *Alnus*, and *Ceanothus*) species form a mutually beneficial relationship with certain bacteria or actinomycetes that convert atmospheric nitrogen into ammonium nitrogen (see fig. 6). This fixed nitrogen is released into the roots of host plants, thereby increasing nitrogen concentrations in living tissue (Tarrant and Trappe 1971). As nitrogen is returned to the soil by litterfall and washing of leaves by rain, other species benefit, such as commercially important conifer species.

"Free-living" nitrogen fixation by bacteria living in close association with roots and mycorrhizae was first suggested by Richards and Voigt (1964), and the phenomenon is now well established (Chartarpaul and Carlisle 1983; Dawson 1983; Florence and Cook 1984; Li and Hung 1987). Nitrogen-fixing bacteria were observed in close association with the mycorrhizosphere of *Pinus radiata* D. Don (Rambelli 1973). We have also found nitrogen-fixing bacteria to associate with ectomycorrhizae of forest trees in the Pacific Northwest (Li and Hung 1987). Bacteria in the genera *Azotobacter*, *Azospirillum*, and *Clostridium* fix nitrogen under conditions characteristic of the rhizosphere, where oxygen concentrations are low (Giller and Day 1985). The amount of rhizosphere nitrogen fixation differs by mycorrhiza type, host plant, and plant community. For example, Amaranthus and others (1989) found significantly higher nitrogenase activity in Douglas-fir rhizospheres associated with cleared areas of *Arctostaphylos viscida* shrubs compared to Douglas-fir grown in adjacent cleared areas of annual grass. Nitrogen is fixed by free-living bacteria in buried wood (Jurgensen and others 1984), an important niche for mycorrhizal development (Harvey and others 1987). Although the amount of nitrogen added to forest sites by free-living microoganisms is small compared to symbiotic sources, steady accretions especially in the immediate root zone can contribute significantly to the overall long-term nitrogen budget. Thus, silvicultural use of rhizosphere organisms to improve nitrogen levels deserves continuing research.

SOIL FAUNA

Many soil animals interact in the rhizosphere. Recent attention has been devoted to soil animals that influence nutrient cycling by acting as microbial grazers in the rhizosphere (Coleman and others 1984; Elliot and others

Figure 6—Scanning electron micrograph of nitrogen-fixing nodules formed on *Alnus sinuata* roots. (Magnification = 7.7x.) (Photo courtesy of Drs. H. B. Massicotte and R. L. Peterson, University of Guelph, Ontario, Canada.)

1980; Ingham and others 1985). Nematodes, protozoa, amoebae, and microarthopods graze on fungi and bacteria in the rhizosphere and release nitrogen in a form available to plants. Nutrient fluxes due to grazing can be significant. Persson (1983) estimated that 10 to 50 percent of total nitrogen mineralization in a Swedish pine forest is mediated by soil invertebrates. Louisier and Parkinson (1984) estimated that testate amoebae alone consume more than 13,000 kg/ha/yr in an aspen woodland soil; 85 percent of that consumption was released or respired. This would release from 25 to 50 kilograms of N/ha/yr from bacterial biomass, an amount roughly equivalent to the amount annually taken up by trees. Invertebrates, particularly arthropods, may also move fungi, bacteria, and other microbes from rhizosphere to rhizosphere.

PATHOGENS AND RHIZOSPHERE INTERACTIONS

The ecology of soil pathogens and pathogen protection by rhizosphere organisms is poorly understood in forest soils; most research has concentrated on the root rot fungi in the genera *Phellinus*, *Armillaria*, and *Fomes,* which infect and persist in large structural roots and stumps. Feeder root pathogens are less well known. Soil biologists hypothesize that a "healthy" forest soil supports populations of microorganisms that compete or otherwise antagonize fine-root pathogens. For example, common pathogens in tree nursery soils are rarely isolated from forest soils. This phenomenon of naturally "suppressive soil" is a subject of considerable research. Ectomycorrhizal fungi can protect trees against fine-root pathogens by (1) providing a physical barrier (fungal mantle) against penetration, (2) depriving root pathogens of carbohydrates, (3) secreting inhibitory antibiotic substances against pathogens, (4) promoting other rhizosphere organisms that inhibit pathogens, and (5) inducing biochemical changes in root cortical cells that inhibit pathogen infection and spread (Marx 1972; Zak 1964).

How do these mechanisms work in forest soils? In Australia, Malajczuk and McComb (1979) found significant differences between rhizosphere populations around mycorrhizal and nonmycorrhizal *Eucalyptus* seedlings in soils suppressive or conducive to the fungal pathogen *Phytophthora cinnamoni* Rands. High counts of bacteria were present throughout the fungal mantle within and between root cortical cells of mycorrhizal seedlings but were not present in nonmycorrhizal seedlings; in culture, many of the bacteria strongly antagonized root pathogens (*Phytophthora* and *Pythium* spp.). In the Pacific Northwest, Rose and others (1980) found that a free-living *Streptomycete* species from the rhizosphere of snowbrush (*Ceanothus velutinus* Dougl.) antagonized three common root pathogens: *Phellinus weirii* (Murr.) Gilb., *Fomes annosus* (Fries) Karst., and *Phytophthora cinnamoni*. In bareroot nurseries, the ectomycorrhizal fungus *Laccaria laccata* (Scop.:Fr.) Berk. and Br. can reduce incidence of *Fusarium* root rot (Sinclair and others 1982). Nonmycorrhizal fungi can also inhibit pathogens. Common soil fungi in the genus *Trichoderma* can reduce the incidence of root rot in pine seedlings (Kelly 1976). Root-protecting organisms, such as symbionts and nitrogen fixers, are important components of "forest health." A research focus on the organisms involved and the effect of management practices on their survival and function is much needed.

SOIL STRUCTURE

An often-overlooked function of soil organisms is their dynamic contribution to soil structure, particularly aggregate formation and stability (Perry and others 1987). The resulting porosity, essential for movement of air and water required by roots and microorganisms, greatly influences forest productivity. Mycorrhizal fungi and other rhizosphere microbes influence soil structure by producing humic compounds (Tan and others 1978), accelerating the decomposition of primary minerals (Cromack and others 1979), and secreting organic "glues" called extracellular polysaccharides (Sutton and Shepard 1976; Tisdale and Oades 1979). Extracellular polysaccharides are especially efficient at stabilizing soil structure and act by linking mineral grains, homogenous clays, and humus into stable aggregates that maintain porosity (Toogood and Lynch 1959).

Because extracellular polysaccharides are also degraded by microbial activity, maintenance of soil structure depends on the relatively continuous flow of photosynthate into the rhizosphere. Without energy flowing from plants to rhizospheres, soil structure may be altered. Borchers and Perry (1989) found that the proportion of large aggregates was significantly reduced in two unreforested southern Oregon clearcuts. The management implications of this important soil biological function are clear: rapid tree regeneration or recolonization by pioneering vegetation are essential to supply soil organisms with the energy needed to maintain a functioning soil structure.

NUTRIENT CYCLING

We typically consider the saprophytic microflora, the decomposers, as the primary soil organisms controlling nutrient cycling. Recent studies, however, show rhizosphere organisms also to be important in nutrient cycling, even though they receive energy primarily from root exudates; for example, ericaceous mycorrhizal fungi and ectomycorrhizal fungi can possess enzymes capable of degrading organic nitrogen into usable forms (Read 1987). Some ectomycorrhizal fungi can also degrade organic carbon sources, albeit at rates typically lower than saprophytic fungi. Under some circumstances, mycorrhizal fungi may compete with saprophytes to slow down overall rates of decomposition (Gadgil and Gadgil 1975). Other direct physiological mechanisms suggest further roles of mycorrhizal fungi in nutrient cycling. For example, ectomycorrhizal fungi release enzymes that increase the availability of phosphorus to higher plants (Alexander and Hardy 1981; Ho and Zak 1979; Williamson and Alexander 1975). This extraction process extends to other nutrients, especially immobile heavy metals, and enters them into the forest nutrient cycle. Many ectomycorrhizal fungi and rhizosphere bacteria produce chelating agents called siderophores that are especially important for iron uptake by plants (Graustein and others 1977; Perry and others 1982; Powell and others 1980). Still other fungi produce oxalic acid that enhances the primary weathering of soil particles (Cromack and others 1979).

Maintenance of forest productivity requires not only the steady cycling of nutrients but also the conservation of nutrient capital. The living microbial biomass in the rhizosphere is exceedingly large, especially when one considers the extensive mats of ectomycorrhizal fungi composed of ropelike hyphal aggregations that tenaciously store nutrients. Thus, few nutrients leach out when populations of rhizosphere organisms are healthy and active. This is particularly important for soluble forms of nitrogen such as nitrate, which is susceptible to leaching. As a primary ecosystem function, rhizosphere organisms form a web to capture and assimilate nitrogen and other nutrients into complex organic compounds and then slowly release them into the forest ecosystem.

RHIZOSPHERE DYNAMICS AND FOREST COMMUNITY DEVELOPMENT

Several exciting directions of rhizosphere research emphasize that the successional dynamics of plant communities and rhizosphere microorganisms are intricately related and interdependent. Because tree harvest and site preparation set the stage for forest community succession, they likewise impact the belowground successional dynamics. We expect that major management implications on forest recovery will develop from soil biological investigations.

Most leads on rhizosphere-plant community dynamics deal primarily with mycorrhizal relationships, so we emphasize those here. Many different host species can form mycorrhizae with the same fungi. Laboratory studies indicate that plants connected by a shared fungus can exchange nutrients, including carbon and minerals, via the "hyphal bridge" (Read and others 1985); if one of the plants is shaded, carbon may move from the strongly photosynthesizing plant in full light to the shaded plant. The implication of this to interactions between plants and nutrient cycling is enormous. For example, understory plants or late successional plants may be able to receive carbon from overstory plants. Seedlings may survive in the shady understory due to overstory carbon support. But another more important mechanism likely occurs in these rhizosphere relationships between plants, a phenomenon termed rhizosphere "legacy" (see Perry and others 1987). A plant species entering in the later stages of succession can grow into a community of rhizosphere organisms that has been developed and maintained by earlier plant species. The later successional plants can utilize a fully functional rhizosphere community developed at the expense (that is, photosynthates) of early successional plant species. Such a functional rhizosphere connection in "successional time" is an important but poorly understood component of community development.

One should not suppose, however, that all plants are connected by mycorrhizal fungi or exchange nutrients through such fungi. Trees that form only ectomycorrhizae cannot associate with VA or ericaceous mycorrhizal fungi, and so plants that exclusively form these different types of mycorrhizae would not be directly linked. Also, several ectomycorrhizal fungi form mycorrhizae only with a particular genus of trees. For example, many fungus species are "host-specific" to Douglas-fir (*Pseudotsuga menziesii*) or the genus *Pinus*. The important point is that such a rich assemblage of rhizosphere organisms with different plant compatibilities enables the soil organisms to "partition" use of the soil by the plants. In essence then, when there is a variety of plants in a forest a related mosaic of belowground rhizosphere organisms partition the soil resource, at times influencing plant-plant interactions and at other times not. We believe that compatible and competitive interactions between plants as mediated by rhizosphere organisms contribute significantly to community development (Perry and others 1989).

One final concept of rhizosphere development is important to forest community succession. Studies in birch (*Betula*) stands in Great Britain reveal that certain mycorrhizal fungi dominate tree root systems of young birch trees, and other fungi are common in older stands (Mason and others 1987). Such fungi have been termed "early and late-stage" fungi, respectively. These two groups of fungi differ physiologically, with the late group generally better adapted to deal with soils high in organic matter. We are only beginning to evaluate such phenomena among mycorrhizal associations in Pacific Northwest forests, where tree and habitat diversity is greater than in Great Britain. We hypothesize, however, that the species composition of rhizosphere organisms shifts tremendously as forests mature, reflecting changes in soil characters as well as forest composition (Amaranthus and Perry 1987). Many ectomycorrhizal fungi show strong preference for specific soil microniches. For example, some fungus species predominate in highly organic substrates such as buried wood; others proliferate in exposed mineral soils. In addition to improving our knowledge on the dynamics of belowground biological succession, understanding such fundamentally

different ecological strategies of mycorrhizal fungi and other rhizosphere microorganisms will allow us to maintain or, if necessary, reestablish viable populations of beneficial soil organisms on degraded sites.

Our studies in the Siskiyou Mountains of southwestern Oregon and northern California provide invaluable leads in testing community-level hypotheses of interactions between plants and soil organisms. Many of the pioneering hardwood shrubs and trees of these forests form mycorrhizae with the same fungi as do the timber tree species. Amaranthus and Perry (1989) show the lasting beneficial "legacy" effect of rhizosphere organisms in the soil beneath pioneering hardwoods. They hypothesize that the diverse plant species in these typically harsh forest habitats have evolved mutual compatibilities between rhizosphere organisms. This ensures not only rapid occupancy of the sites by pioneering plants but also maintenance of soil organisms that will benefit later successional stages. Such a legacy may also operate under the proposed system of green tree retention; the ramifications of such rhizosphere mechanisms may be important in other forest areas in the Pacific Northwest.

CONCLUSIONS AND FUTURE DIRECTIONS

As we develop holistic approaches to understanding forest ecosystems and integrated, ecologically based management tools, we must factor in the inseparable connections to soil organisms. Just as forests invest tremendous capital in the form of photosynthates to fuel beneficial soil organisms, so too must we protect this unseen and overlooked ecosystem. We need to better understand its "functional biodiversity."

The number of species of microorganisms in the soil is far greater than aboveground plants and animals, but it is difficult to quantify. Also, although many soil organisms perform similar processes, reflecting a "redundancy in function," they may have different ecologies; for example, different functions during the year or over successional time. Our goal was to define and characterize viable population levels of critical functional groups in a diversity of forest types and age classes so that we can predict when forest systems are becoming degraded. Understanding how the functional biodiversity of the soil biota acts as a biological "buffer" to forest disturbance and contributes to recovery will be an area of intense investigation.

Other challenges involve linking soil biological science to the complex issues of long-term site productivity and vegetation shifts during rapid global climate change. Sharpened understanding of soil biology, particularly the tight linkages in the rhizosphere, is paramount to developing sound ecological guidelines to protect the living resource of the soil.

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SOIL MANAGEMENT AS AN INTEGRAL PART OF SILVICULTURAL SYSTEMS

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ABSTRACT

Forest management is at a critical juncture. Concepts and strategies for managing forests to produce goods and services, yet maintain functioning visually pleasing forests, are being debated, developed, and implemented. A well-designed and implemented silvicultural system is basic to good forest management. Means of protecting soil and all factors affecting soil properties must be integrated into silvicultural systems. Inappropriate silvicultural techniques can degrade forest productivity especially by compacting, displacing, or destroying soil surface layers rich in organic matter. Because both short- and long-term productivity can be influenced by changes in these layers, we make several recommendations on how to protect soil when developing prescriptions for any silvicultural system.

INTRODUCTION

Forest management is at a critical juncture. Not since the late 1960's and early 1970's has the management of forests been more scrutinized. The management controversies of the Bitterroot and Monongahela National Forests contributed to passing of the National Forest Management Act of 1976. In response to the Act, forest practices were modified, but the primary focus of forest management continued to be on the production of timber-based commodities. Silvicultural systems were designed to efficiently produce lumber and fiber-based products from a variety of lands. Much of society was content with this form of management until more and more virgin forests were converted to young, fast-growing, high-yield, commodity-producing tree farms.

The demands of society are changing as the 21st century approaches. No longer is commodity production the only desirable forest attribute. Diversity, sustainability, old-growth, scenic values, wildlife (both game and nongame), and water quality are becoming increasingly important. To provide and maintain these attributes requires some basic changes in forest management; new initiatives and concepts are now being developed.

"New Forestry" is one of these. It is a concept defined as "the management of ecosystems and not just the regeneration of trees" (Franklin 1989). In addition, a broad philosophy of forest management by the Forest Service, U.S. Department of Agriculture, entitled "New Perspectives," is also in the process of being defined and implemented. Both of these strive to make forest management more broadly based ecologically and responsive to society's demands.

New perspectives and new forestry both use the ecosystem as the basic unit of land management. The management of forest ecosystems to meet objectives is the practice of silviculture (Smith 1962). The silvicultural methods and manipulation techniques available to manage forests need to be assembled into complete silvicultural systems, planned programs of treatments to be applied throughout the life of the forest. This will be more important than ever, if new forestry and new perspectives are going to succeed and if ecologically sound silvicultural systems are to be implemented. Soil is often not fully integrated into our silvicultural systems, yet as outlined here, it is critical to the regeneration, productivity, nutrient value, and moisture-retention abilities of all forested sites. After discussing the role of surface soil layers in the forest, we outline silvicultural methods and their effects on soils, and recommend soil protection guidelines for silvicultural prescriptions.

FOREST SOILS

Litter, humus, soil wood, and certain key properties of the surface mineral layers of forest soils are usually most critical when developing silvicultural systems. These are the soil layers most easily and commonly disturbed by silvicultural activities, yet they are crucial to forest productivity.

Woody Residues

Although they are not a soil component, the quantity, quality, and disposition of woody residues can also influence forest soils greatly. The quantity of downed material can vary dramatically, depending on site, forest conditions, and forest treatments (table 1).

Physically, woody residues protect soil from erosion, displacement, and compaction. In addition, they can protect regenerating forests from both abiotic and biotic elements

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Table 1—Average downed woody loadings, duff depths, and percentages rotten from the Forest Inventory by cover type for east and westside National Forests of the Northern Rocky Mountains (Brown and See 1981)

	Westside forests			Eastside forests		
Cover type[1]	Total	Duff depth	Rotten large	Total	Duff depth	Rotten large
	Tons/acre	*Inches*	*Percent*	*Tons/acre*	*Inches*	*Percent*
PP	12.9	0.6	61	5.0	0.6	70
DF	15.7	.9	63	12.0	1.0	63
LP	17.5	1.1	59	16.1	1.1	55
L	21.4	1.2	58	—	—	—
S-F	26.7	1.4	52	22.7	1.3	52
C-H	33.4	1.4	57	—	—	—

[1]PP = Ponderosa pine, *Pinus ponderosa*
DF = Douglas-fir, *Pseudotsuga menziesii*
LP = Lodgepole pine, *Pinus contorta*
L = Larch-grand fir, *Larix occidentalis-Abies grandis*
S-F = Spruce-subalpine fir, *Picea engelmannii-Abies lasiocarpa*
C-H = Cedar-hemlock, *Thuja plicata-Tsuga heterophylla.*

(Edgren and Stein 1974). Residues provide shade and protection from wind and snow, and can be critical factors in protecting newly established seedlings from livestock (Graham and others, in press). Regeneration can be damaged by falling or rolling logs, and fuel loadings can threaten forests by becoming a fire hazard. As logs and other woody debris decay they can become incorporated into the litter, humus, and mineral soil horizons.

Chemically, woody residues are an important forest component. Decaying logs, especially those with the incipient and advanced forms of decay, are excellent substrata for nonsymbiotic nitrogen (N) fixation (table 2). If sites do not have nitrogen-fixing plants present, such as ceanothus (*Ceanothus* spp.), shepherdia (*Shepherdia canadensis*), or leguminous forbs, these nonsymbiotic forms of nitrogen fixation are crucial. Also, woody residues release nutrients during decay that are critical for forest growth (Harvey and others 1987). Therefore, the quantity and kind of forest residues on a site can be critically important for maintaining site productivity.

Litter

The surfaces of forest soils are usually covered with a layer of litter. This material is composed of organic matter (OM) from trees, shrubs, grasses, forbs, other plant material, and animal material from the site. This layer is unconsolidated and undecomposed. It can vary widely depending on site, decomposition rates, and vegetation (table 3).

Litter layers protect the surface of the soil by acting as a mulch on the soil surface to retain moisture in the lower layers. Soils protected by litter are less prone to erosion (Rothacher and Lopushinsky 1974). Likewise, litter layers protect soils from compaction (Lull 1959). Litter is not usually an important site for ectomycorrhizal activity, but if moisture is maintained in these layers ectomycorrhizae are present (Harvey and others 1987).

Litter can be an impediment to both natural and artificial regeneration of trees. If the litter layer is thick, which often occurs in ponderosa pine (*Pinus ponderosa*) forests where decomposition is slow, tree seeds falling on this layer germinate, but because litter dries rapidly they may die (Pearson 1949). On the other hand, if litter remains moist during germination and early seedling growth (a situation that can be found in many hemlock [*Tsuga*] and cedar [*Thuja*] sites) successful establishment of tree seedlings is quite common (Minore 1972). Often the forest being generated on these organic seed beds does not produce the desired mix of species or stand structure to meet some forest management objectives.

Humus

Humus is made up of highly decomposed organic material with no recognizable plant parts. Humus layers are usually relatively shallow, depending on the site, decomposition rates, and vegetation (table 2). Humus is a dynamic part of the soil horizon. It is a major substrate for both high nitrogen fixation and nitrogen storage, and it is rich in other essential elements (calcium, potassium, magnesium) (Harvey and others 1987). Humus contains many roots and maintains soil moisture, making it important for ectomycorrhizal activity. This layer often represents the

Table 2—Woody residue weights[1] and associated N additions[2] from nonsymbiotic N fixation on four old-growth forest sites in western Montana and northern Idaho (Jurgensen and others 1987)

Decay class	Douglas-fir (Montana)		Cedar-hemlock (Montana)		Subalpine fir (Montana)		Cedar-hemlock (Idaho)	
	Tons/acre	*Lb/acre*	*Tons/acre*	*Lb/acre*	*Tons/acre*	*Lb/acre*	*Tons/acre*	*Lb/acre*
Incipient	5.0	0.03	10.7	0.07	27.4	0.17	3.1	0.04
Intermediate	7.1	.12	15.5	.18	15.7	.26	21.3	.40
Advanced	8.0	.20	10.9	.25	21.8	.70	44.4	2.71
Total	20.1	.35	37.1	.51	65.0	1.14	68.8	3.15

[1]Dry weights of woody residues >3 inches in diameter.
[2]Total N fixed over a 180-day period.

Table 3—Depth and distribution of soil organic materials among major soil components (Harvey and others 1989)

Site description	Organic matter depth[1]	Percent distribution in: Litter	Humus	Decayed wood
	cm			
Undisturbed stands:				
Hemlock-climax (MT)	3.8	12	38	51
Subalpine fir (MT)	3.5	7	45	48
Douglas-fir (MT)	2.3	6	58	35
Ponderosa pine (WA)	2.0	30	19	51
Grand fir (ID)	1.7	31	68	2
Hemlock (ID) 200-yr WWP[2]	1.5	25	61	14
Disturbed stands:				
Subalpine fir (MT) 50-yr LPP[2]	1.9	19	58	23
Subalpine fir (MT) 15-yr WL[2]	1.5	21	41	39
Subalpine fir (MT) 15-yr LPP[2]	.5	46	40	14

[1]Thickness of organic matter layers in the surface 30 cm of soil.

[2]Major conifer species occupying the site, WWP = western white pine (*Pinus monticola*), LPP = lodgepole pine (*Pinus contorta*), WL = western larch (*Larix occidentalis*).

transition between organic layers and the mineral soil. Very often this layer when burned over can provide numerous microsites for natural regeneration (Haig and others 1941).

Soil Wood

Soil wood as a component of forest soils is often overlooked. Soil wood is highly decomposed wood, incorporated into soil horizons usually in the form of brown cubicle rot. It can also occur in a fine powdery form, usually much older. As with the other organic components, quantity and type can be quite variable (table 2). Soil wood is a dynamic part of the soil system. Soil wood is not a continuous layer, but occurs in deep pockets created by buried logs and decaying stumps (Reinhardt and others, in press). Soil wood should not be confused with rotten wood or residue that is on the surface of the soil.

Soil wood protects soils from compaction and provides OM to the mineral soil. It is also an important source of N fixation and N storage (Harvey and others 1989). Soil wood is an excellent substrate for ectomycorrhizae. In addition, because of its water retention ability and physical characteristics, soil wood often contains root systems of conifers that grow rapidly and concentrate in these buried logs and stumps.

Surface Mineral Soil

The surface 5 to 10 cm of mineral soil is derived from the parent materials of the site, but is also highly influenced by vegetation and surface organic layers. The OM incorporated into shallow mineral horizons carries important properties into the mineral soil base. Mineral soil with good OM levels has better nutrition, water-holding capacity, and structure than soils with small amounts of OM. In addition, OM-rich mineral soils are excellent sites for nitrogen fixation and ectomycorrhizae (Harvey and others 1987). Surface layers are highly susceptible to compaction and displacement. Mineral soils with high volcanic ash content are particularly sensitive to forest operations. (See Hironaka and others, Page-Dumroese and others, these proceedings).

THE SILVICULTURAL SYSTEM

A silvicultural system is a planned program of treatments applied throughout the life of a stand (Smith 1962). The silvicultural system should integrate all planned site treatments and methods that will be used to implement the system (Graham 1990). Numerous factors should be considered during its development. Foremost among these are the site and stand conditions that determine the timing of treatments and what treatments are required to meet management objectives.

Regeneration Methods

The choice of regeneration method is, in many ways, the most critical decision regarding the entire system. It should be selected carefully and consider all abiotic and biotic elements that might influence forest regeneration and development, but soil properties are especially important. Physical properties of the mineral soil, especially water-holding capacity, can be used to help determine which regeneration method would be the most appropriate (fig. 1). For example, stands located on soils with high amounts of available water would be most suited to the clearcut method and stands on droughty soils with little available water would be more appropriate for selection or heavy shelterwood methods.

To implement the regeneration method, timber harvesting and harvesting machinery are usually used. Forest managers, for the most part, understand the importance of protecting forest soils from erosion and compaction through the application of careful harvesting techniques dependent on soil type and slope of the harvest unit. Even

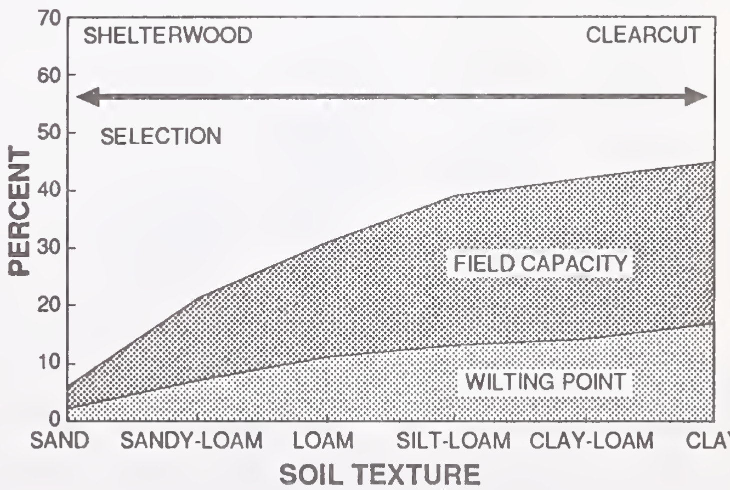

Figure 1—Regeneration methods in relation to soil moisture and texture.

with this awareness, however, the importance of minimizing total soil disturbance is not always recognized.

In general, cable yarding systems compact and displace less soil than tractor yarding systems. Also, by using cable yarding over snowpacks during the winter, for example, on extremely shallow and sensitive soils, even less soil disturbance occurs (fig. 2). Ground-lead yarding in the summer can cause considerable soil disturbance. This can be very destructive to the critical surface layers of organic matter and shallow mineral soil.

Tractor yarding not only disturbs more soil than cable yarding, it also can compact soil. Compaction occurs not only in the surface layers, but at greater soil depths (Froelich and others 1985). Soil compaction in deep soil layers can be greater and more long-lasting than at the surface because of deep-layer isolation from organic matter and surface litter. Because yarding techniques can displace and compact forest soils, techniques specified in the silvicultural system need to be appropriate for the soils on which they are to be used. Care should always be taken to minimize soil displacement and compaction during any harvesting operation.

As harvesting equipment and processing plants become more efficient, and wood utilization becomes more intensive, greater amounts of fiber are removed from the forest. By removing this material less organic material may be available for incorporation into the soil. Therefore, it will become more critical that sufficient organic material remains after intensive harvesting to provide organic parent material to maintain both short- and long-term productivity.

Postharvest Treatments

After the choice of regeneration and harvest methods, fire hazard reduction and site preparation become the most important elements of the silvicultural system. Both natural and artificial regeneration require creating the proper conditions for establishment and growth, usually a compromise. Objectives of site preparation include: exposing mineral soil, creating access for planters, reducing competing vegetation, reducing the incidence of disease, reducing wildfire hazard, minimizing seedling damage, and encouraging early growth. Often when meeting these objectives the factors critical for tree growth (moisture, temperature, light, and nutrition) are not optimized (Graham and others 1989a).

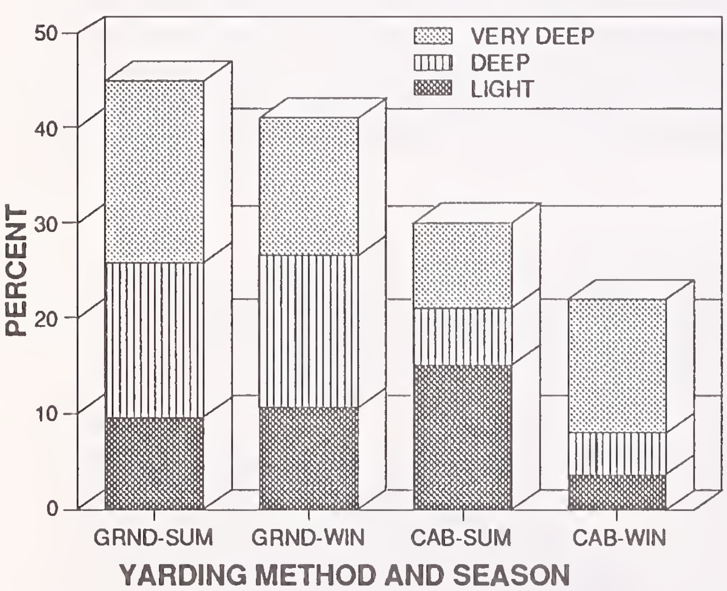

Figure 2—Soil disturbance by depth class (light 0-2 inches, deep 2-10 inches, very deep >10 inches) for ground (GRND) yarding during the summer (SUM), ground yarding during the winter (WIN), cable (CAB) yarding during the summer, and cable yarding during the winter for harvesting operations in British Columbia (Krag and others 1986).

Specific soil characteristics need to be included in system planning. How much or how little soil disturbance is necessary to establish and grow specific forest vegetation? Depending on the forest floor depth and condition of residue, litter, humus, and soil wood, highly variable disturbances may be required for artificial regeneration. Trees can be planted successfully near and through surface organic materials as long as good root-to-soil contact is made and the medium is moist. Likewise, depending on the desired vegetation, very little disturbance may be needed for natural conifer regeneration.

There are three methods of preparing sites for regeneration: mechanical, chemical, and fire. Chemical methods of site preparation are infrequently used in the Inland and Pacific Northwest. When they are used, a good understanding of how they interact with the soil is extremely important. Depending on site and soil characteristics, chemicals can be volatilized, absorbed, leached, or degraded. Therefore, chemical site preparation should be carefully applied and fully integrated into the silvicultural system when it is used (Baumgartner and others 1986).

Tractor piling of logging debris and machine site preparation is used widely throughout the West. Similar to tractor yarding, tractor piling can both compact and disturb soil surface layers (Minore and Weatherly 1988, 1990). The more the disturbance, the greater the loss of surface organic layers. Because tractor piling does impact the soil layers through displacement and compaction, tractor piling of logging debris can adversely affect tree growth (Bosworth 1989). For example, of several young stands measured in western Oregon the majority were growing below their potential after the sites were tractor piled (fig. 3). Likewise, when organic horizons are maintained or enhanced, seedling growth can be improved and long-term site productivity increased (Graham and others 1989b, in press).

Broadcast burning, if properly applied, can be highly effective and beneficial to the site (Graham and others 1989a). If burning conditions are optimum when the humus and soil wood are moist and fires do not destroy these materials, nutrients, especially N, can condense in underlying mineral layers when released from the burning litter and other surface materials (Harvey and others 1989). Broadcast burning, if properly applied, can allow stands to grow closer to their potential than tractor-piled stands (fig. 4).

After the site preparation and hazard reduction treatments are completed it is imperative to leave sufficient large woody material on the site. As mentioned earlier, residue has several properties important for maintaining forest productivity. We recommend leaving a minimum 10 to 15 tons per acre of large woody material (>3 inches

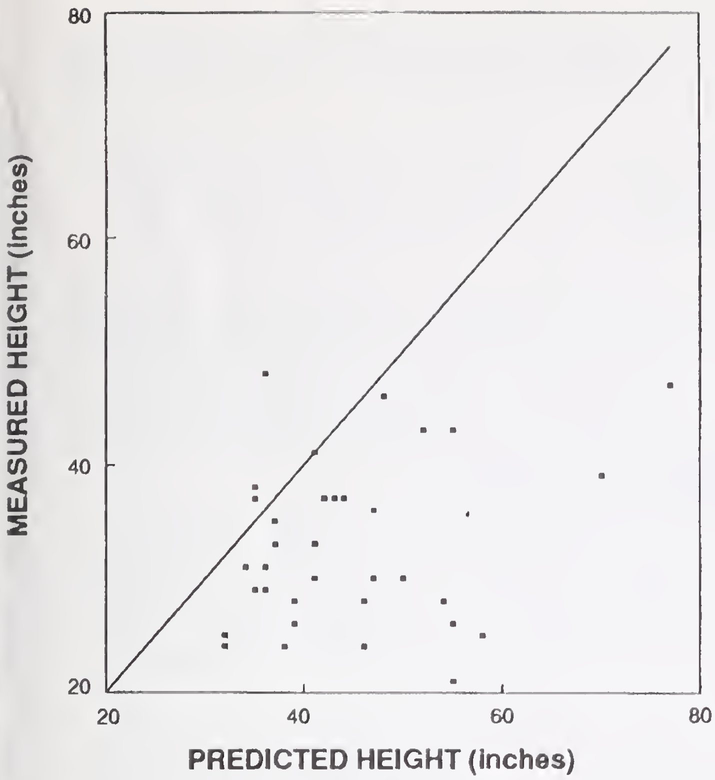

Figure 3—A comparison of measured Douglas-fir seedling heights on plantations where slash was broadcast burned. Points below the diagonal line indicate measured heights shorter than predicted heights at age 5. Points above the line indicate measured heights taller than predicted heights (Minore 1986).

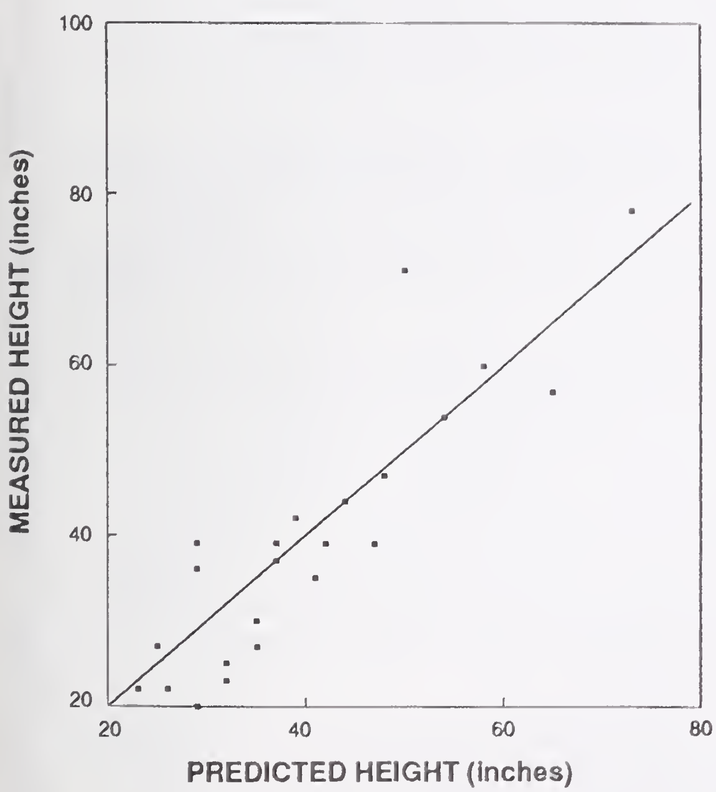

Figure 4—A comparison of measured Douglas-fir seedling heights on plantations where slash was piled and burned. Points below the diagonal line indicate measured heights shorter than predicted heights at age 5. Points above the line indicate measured heights taller than predicted heights (Minore 1986).

in diameter) after timber harvesting and other site treatments (Harvey and others 1987). We are presently refining these recommendations to make them more site specific based on habitat type.

Intermediate Stand Treatments

Cleanings, weedings, and thinnings are common intermediate stand treatments used in silvicultural systems. If these operations are mechanized, there is potential to compact and displace soil. Consequently, the same concerns described for harvesting and site preparation treatments, in relation to soil properties, should be evaluated when applying intermediate treatments. Another concern with intermediate treatments is the removal of material that normally would be added to the organic component of soil. Removal of wood material during intermediate treatments and final harvest should be limited and always allow for sufficient woody debris to remain on site to comply with the recommended amount needed to maintain long-term productivity.

RECOMMENDATIONS

We have several specific recommendations on how to protect and manage soil when developing prescriptions for any silvicultural system.

1. Consider soil properties such as texture, nutrition, depth, structure, and organic matter content, in all phases of silvicultural system development.

2. Be vigilant in protecting the surface OM and mineral layers.
   a. Use cable systems for yarding when possible and time these operations to minimize soil disturbance.
   b. Design tractor yarding systems to minimize number of trails and disturbance and do not operate under high moisture conditions.
   c. Use tractor piling of logging debris judiciously. Grapple piling of debris is preferred.
   d. Use prescribed fire when possible for fuel reduction and site preparation where necessary. Maintain high moisture content in humus and soil wood when burning.

3. Prescribe site preparation to meet specific biological objectives for the development of forests and not to meet administrative objectives.

4. Make sure that minimum woody residues are maintained on site (10 to 15 tons/acre until more site-specific recommendations can be developed). This is critical as wood utilization intensity is increased.

5. Minimize the destruction of either incipient or advance decay forms of residues.

6. Do not allow excess fuel accumulations, but protect ecosystems from late-summer wildfires.

CONCLUSIONS

Soil is literally the foundation of ecosystems. It must be protected if forests are going to be sustained and is even more critical when the threats of global climate change and

increased acid deposition are contemplated. As soil is the foundation of the ecosystem, the silvicultural system is the foundation of forest management. Therefore, means of protecting soil and all of its properties must be integrated into the silvicultural system if the concepts of new forestry and new perspectives are going to succeed.

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Soil Productivity

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CONNECTING SOILS WITH FOREST PRODUCTIVITY

Dan Binkley

ABSTRACT

The productivity of Rocky Mountain forests is lower than forests in most other regions due to shorter growing seasons and low precipitation. Nutrient availability also appears to limit most forests in the region. Although operational fertilization remains uncommon in the region, standard forest management practices have large impacts on soils that can increase or decrease nutrient availability. Sustained productivity cannot be assured without improving our understanding of the effects of management on soils.

INTRODUCTION

We all know that soils are a crucial factor in the productivity of forests, and that forest management practices alter soil properties that in turn alter forest productivity. Extreme cases are perfectly clear; fast-growing plantations of *Eucalyptus* in Brazil cannot be sustained without frequent fertilization. What about the Rocky Mountain region, where short growing seasons, cool temperatures, and dry soils conspire to limit growth? Surprisingly little work has focused on the connections between soils and forest productivity in the Rockies; available evidence demonstrates strong, reciprocal relationships between forests and soils.

Connections between soil nutrient availability and forest growth have been well established in the Pacific Northwest and Southeastern United States, where each year thousands of hectares are fertilized with N and P to compensate for limiting soil supplies (Allen 1987; Binkley 1986). To what extent does the availability of soil nutrients limit growth in the Rocky Mountain region? Only a handful of studies have examined nutrient limitations in this region, but all have found that nitrogen availability limits growth on at least some of the sites examined (table 1). Conclusive evidence for limitation by other elements is less common, but indications of responses to fertilization with P and K have been reported.

Low rates of forest productivity limit the investments that are warranted in forest management; profitable opportunities for fertilization may be limited in this region to late-rotation periods. Given limited interest in intensive management and fertilization, how much interest is warranted in soil nutrient availability? Sustained-yield forestry probably cannot be achieved in the Rockies without careful management of stand nutrition. Operations such as harvesting and burning remove nutrients and alter the rates at which nutrients recycle. If current rates of forest production are nutrient limited, such changes are likely to change future productivity.

Table 1—Demonstrated nutrient limitations in the Rocky Mountain region

Species	Nutrient limitation
Lodgepole pine	N, P?, K? (Weetman and Fournier 1982) N , P, S? (Yang 1985a) N , S, P? (Yang 1985b) N (Waring and Pitman 1985) N (Hunt and others 1988) N, P (Weetman and others 1988) N?, P?, S? (Cochran 1989) N, P?, K? (Binkley and others 1990)
White pine	N, P?, K? (Loewenstein and Pitkin 1963)
Ponderosa pine	N (Heidmann and others 1979) N, P? (Cochran 1979)
Douglas-fir	N, K? (IFTNC 1989) N (Shafii and others 1989)
Grand fir	N, P?, K? (Loewenstein and Pitkin 1971) N (Shafii and others 1989)

IMPLICATIONS OF SIMPLE CALCULATIONS

Some implications become clear from simple calculations. For example, a wide range of studies have shown that the amount of nitrogen lost in fires is directly proportional to the amount of organic matter consumed; about 5 kg of N are lost for every 1,000 kg of biomass burned (Binkley and Christensen 1991). A typical slashfire might consume about 50,000 kg/ha of fuel, reducing the capital of nitrogen by 250 kg/ha. How important is this amount of nitrogen? Two comparisons are enlightening. A stem-only harvest of a typical Rocky Mountain forest would remove about 60 to 150 kg/ha of nitrogen (Clayton and Kennedy 1985; Stark 1982), so a typical slash fire could remove as much nitrogen as the biomass harvested in two or more rotations. The N content of precipitation in the region is about 2 to 4 kg N/ha annually, and free living N fixation (see Jurgensen, this proceedings) probably adds another kilogram or so. At a rate of 3 to 5 kg/ha of inputs annually, a loss of 250 kg-N/ha equals 50 to 80 years of inputs. This might lead a forester to figure that given a 80- to 100-year rotation, the effects of slash burning on

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N loss are quite acceptable. The forester might be right if sustained production were the only goal. However, the forester might take less comfort in that conclusion if she realized the site was N limited, and that the fire prescription prevented a natural increase in productivity that might have accompanied a 250-kg/ha increase in soil N over that period. The fertility of soils is malleable, and the productive capacity of a site at any particular time is likely increasing due to nutrient accumulation, or decreasing due to recent effects of fire or harvest.

Not all forest management operations reduce nutrient availability. In many cases, forest harvest (and even burning) increases the availability of nutrients at a time when the demand for nutrients to expand the canopies of regenerating trees is great. Again, little quantitative information is available for the Rockies, but what little we know is consistent with results from other regions (see Page-Dumroese, this proceedings).

Nutrient availability may be even more fundamental to forest management in the Rockies than these examples would suggest. Yield tables for our region typically show maximum periodic annual increment near age 40 to 80, followed by substantial declines in later decades. This observation of declining growth in relatively young (<100-yr) stands is so common that we typically don't bother to ask why it occurs. Recent work in central Colorado (M. G. Ryan, personal communication) and in southeastern Wyoming (F. W. Smith, personal communication) shows that leaf area of lodgepole pine forests declines substantially after about age 50 to 70, coincident with declines in stem growth. Fertilization trials have shown no response of leaf area in stands younger than 50 years, but substantial increases in leaf area in older stands (Binkley and others 1990). The implications are far reaching: declining nutrient availability may lower stand leaf area, which in turn lowers stand productivity. If such patterns of soil nutrient availability with stand age applied across the region, all our site index estimates and stand yield tables would have built into them the effects of declining nutrient availability (which might result from accumulation of readily available nutrients in noncycling biomass, but we don't know for sure). Any management activity that altered nutrient availability would alter the trajectory of stand yield, and alter such calculations as the culmination of mean annual increment and optimal rotation age.

FOREST EFFECTS ON SOILS

In addition to the role of soils in determining potential rates of forest production, the state of a forest also has reciprocal effects on soils. Forest harvesting can greatly increase the concentrations of nitrate leaching in soil solutions by an order of magnitude or more (Knight and others 1991; Stottlemyer 1987). These postcutting losses amount to only a few kg/ha of N, and are much lower than the responses of N-rich northern hardwood forests (such as Hubbard Brook; Likens and others 1978); nonetheless, they demonstrate dramatic interactions between the soil system and trees.

The state of a forest also has a strong influence on soil moisture in our region. Harvesting a mixed conifer forest in the Fraser Experimental Forest in central Colorado resulted in a reduction in evapotranspiration of about 50 mm/yr, plus a greater snowpack accumulation (due to reduced sublimation of snow held in the canopy) of 150 mm/yr (Troendle and Kaufmann 1987). These hydrologic changes after forest removal result in large increases in soil moisture. For example, Newman and Schmidt (1980) found that soils in a larch/Douglas-fir forest in Montana contained less than half the water (on a weight basis) contained in soils in clearcut sites throughout the growing season. Such changes in soil moisture likely produce large changes in microbial activity and nutrient availability, but such responses remain virtually unexamined in the Rocky Mountain region.

Water losses due to interception and evapotranspiration also differ among species under the same site conditions. For example, Kaufmann (1985) estimated that evapotranspiration in aspen stands removed only about half as much soil water as in lodgepole pine stands, and only one-third as much as in Englemann spruce stands.

Forests also exert considerable control over the temperature regimes at the surface of soils. A classic paper by Hungerford (1980) illustrates the dramatic differences in temperature extremes at the soil surface for different degrees of canopy removal. The low temperature at the soil surface in a clearcut in Montana's Lubrecht Experimental Forest on August 29 of 1978 was -5 °C (23 °F), and the high on the same day was 56 °C (133 °F). Under an intact forest, the low and high were 2 °C (36 °F) and 35 °C (95 °F). High temperatures in the clearcut may be lethal to regenerating seedlings, and the temperature effects on microbial processes are likely substantial (especially coupled with higher soil moisture).

It is difficult to remove logs from a forested landscape without compacting soils, and it is amazing how little we know in the Rocky Mountains about the degree and pervasiveness of soil compaction, the time course of recovery, and the implications for forest productivity (see Froelich, this proceedings). This is perhaps the most critical gap in our understanding of the connection between forest management and the maintenance of forest productivity in our region.

IMPLICATIONS

Soil fertility constrains forest productivity in the Rockies, and forest management activities certainly alter soils. We currently have very little information on the pluses and minuses of common forest practices for this large region of the country. We can continue getting by with the status quo, but that would not be responsible land management.

Two critical needs are obvious. The first is to educate foresters about the fundamental connections between soils and forest productivity (highlighting the particular importance of soil organic matter), which was a primary goal of this workshop. The second is a need for more

research, which sounds predictably familiar from an academic such as myself. However, the most critical sort of research is not esoteric investigations that only a scientist could love. What we need to understand (= conduct research on) is simply what our current management practices are doing to the productivity of our soils and forests (see Powers, this proceedings). How can we be confident that our current practices lead to sustainable (or preferably, increasing) productivity if we have almost no information on the effects on soils?

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Speakers answered questions from the audience following their presentations. Following are the questions and answers on this topic:

Q. (from Hersh McNeil)—Do you think something could be done to increase N availability later in stand development?

A.—Only a few percent of the total quantity of nutrients bound in soil organic matter become available each year for plant uptake; any treatment that could increase this percentage could (1) reduce nutrient limitation of current growth rates, and (2) increase the rate at which nutrients accumulate in harvestable biomass. The most proven approach to increasing turnover of soil nutrients is harvesting; warmer and wetter conditions following harvest often increase nitrogen availability by two-fold (we need to know more about the size and variability of this response). The response of nutrient availability to thinning has not been examined in this region; in many cases, thinned stands respond well to fertilization, which suggests indirectly that any increase in nutrient cycling is slight (and may in fact be negative, if thinning slash allows microbes to compete with plants for N). Prescribed fire is another possibility; W. W. Covington and colleagues demonstrated that prescribed burning in ponderosa pine in northern Arizona resulted in both an immediate increase in soil ammonium, and an increase in the rate of decomposition of the remaining forest floor. Fertilization with one element (such as P) may increase the release of other elements (such as N) through accelerated decomposition, but this hasn't been examined in our region. Finally, the choice of species may be the greatest opportunity for influencing nutrient cycling rates. The inclusion of N-fixing species typically accelerates the cycling of all elements. Less is known about the differences among non-N-fixing species, but studies from the Eastern United States and elsewhere suggest that rates of decomposition and nutrient cycling may differ substantially among species. Unfortunately, no one has examined nutrient availability in replicated plantations of different species in our region. I suspect some of the greatest differences would be found between stands of aspen and conifers; the nutrient-rich litter of the aspen combined with the frequent presence of understory N-fixers probably promotes much faster nutrient cycling under aspen.

ARE WE MAINTAINING THE PRODUCTIVITY OF FOREST LANDS? ESTABLISHING GUIDELINES THROUGH A NETWORK OF LONG-TERM STUDIES

Robert F. Powers

ABSTRACT

Protecting the inherent capacity of forest land to grow vegetation is a goal of modern forest management and a legal requirement of the Forest Service, U.S. Department of Agriculture. Options for judging this capacity include potential tree growth and potential net primary productivity of all vegetation. Evidence exists that this capacity is degraded by substantive losses of site organic matter and soil porosity. If such potentials were known for each site, and if changes in key soil properties were calibrated against these potentials, the relative impacts of forest practices could be assessed objectively and directly. Unfortunately, few data have been collected for establishing such calibrations, and managers must turn to soil-quality monitoring standards that are based more on professional judgment than on rigorous science. This paper describes a cooperative study between Research and Administrative arms of the Forest Service to establish such calibrations through a national network of long-term productivity studies.

INTRODUCTION

Sustaining the long-term productivity of our Nation's forests is an ethical and economic aim of forest management. Ethics relates to a willingness to sacrifice personal gain for the good of a social pyramid with society at its apex and the land at the base (Leopold 1949). Economics refers to protecting the land's ability to meet consumer demands by sustaining a profitable flow of forest products. For the Forest Service, U.S. Department of Agriculture, there is a legal reason, too. Among the World's nations, the United States is unique in its mandate for good stewardship of public lands. By law, the Forest Service must monitor the effects of management practices to ensure sustained productivity.

This paper serves several purposes. First, it reviews the main concepts of forest site productivity. It also acquaints the reader with the legal basis for protecting long-term productivity and its implications for field monitoring. Next, the paper summarizes what is known about productivity decline due to poor management. Finally, it describes a newly established Forest Service cooperative program that tackles the subject head-on.

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SITE PRODUCTIVITY CONCEPTS

The productivity of a forested site is shaped by abiotic factors of climate, soil, relief, and mechanical damage by fire, flood, or wind. Productivity also is shaped by biotic factors involving tree genetics, stand age, stocking and degree of competition, and stand health as conditioned by insects, diseases, other pests, and symbionts. Except for such random disturbances as fire, flood, or wind, abiotic factors change relatively slowly under natural conditions. In contrast, most of the biotic factors (tree age, stocking, health) are dynamic. For example, growth rates vary with age. Usually, they are low during juvenility, accelerate with approaching maturity, and decline thereafter. Some abiotic and many biotic factors can be affected by management—either by design, or inadvertently. And the degree to which this is understood depends on one's definition of "productivity."

Timber Site Quality—Historically, the forestry concept of site productivity has centered on the capacity of the land for growing wood of commercial value. Traditionally, this capacity or "timber site quality" is expressed as volume increment averaged over a period of years. Timber site quality has both an actual and a potential component.

Actual and potential volume increment rarely coincide. The former is the fraction of a site's capability that is actually realized by management. Generally, this is less than the site's potential. For example, trees spaced widely in understocked stands will have lower collective growth rates than trees growing closely. Furthermore, trees weakened through severe and prolonged competition or from mechanical injury are susceptible to insects and diseases, which decreases absolute growth. All of this means that the actual site productivity of a given unit of forest land varies continually. A corollary is that management can affect apparent productivity simply by manipulating stocking, species, and pests.

Besides its actual productivity, each land unit has a theoretical maximum productivity, or an upper limit for

the rate of wood growth when the stand is fully stocked and other factors are held constant. This is the concept of absolute or potential site quality (Spurr 1952), only part of which generally is realized by management. Estimating this potential is difficult. Variable density yield functions may be manipulated to approximate the potential upper limit of wood growth, but empirical yield tables generally reflect norms for stocking and incorporate incipient degrees of stress from insects and diseases. Thus, yield tables would tend to underestimate potential site quality.

Although it is the historical measure of site quality, wood production has a relatively low priority in the tree's allocation of photosynthate. Production of roots, foliage, and reproductive structures all take precedence (Grier and others 1989). Thus, wood production is a rather incomplete and limited measure of the quality of the whole site for forest growth. Is there an alternative?

Total Site Productivity—The fundamental ecological measure of site productivity is total dry matter produced by autotrophic plants. This "net primary productivity" (NPP) is expressed by the equation:

$$NPP = GPP - Rs_A$$

where GPP is gross primary productivity from total photosynthesis and Rs_A is respiration of all parts of autotrophic plants.

Units of NPP generally are expressed as dry mass of organic matter produced per unit area each year, such as Mg/ha/yr. Not all NPP appears as standing biomass. Some that is produced annually is cycled to the forest floor and soil or is consumed by grazing animals. The quantity that does accumulate can be defined as:

$$NPP_B = NPP - NPP_C - NPP_D$$

where NPP_B is the net primary productivity accumulated annually in standing biomass of all plants above ground, below ground, or both; NPP_C is the biomass consumed annually by pathogens, insects, and grazing animals; and NPP_D is the actual production of detritus from litterfall and root mortality.

As indicated by the paucity of estimates of NPP available for forests of western North America (table 1), NPP and its individual components are not estimated easily. Tables and equations for estimating tree stem volumes from diameter and height are common, but these generally have a commercial bias (bole volumes to a minimum top diameter, rather than dry matter) and do not account for branches, foliage, or roots. Usually, destructive sampling, drying, and weighing are used to relate the mass of individual tree components through regression to more easily measured tree dimensions. Measurements taken at different times can then be used to estimate standing NPP. Obviously, such work is extremely costly, and the equations often are specific only to the stand in question. The detritus component (NPP_D) can be estimated by weighings of periodic litterfall, but this can be complicated by storm-induced detritus. Estimating NPP_C is even more difficult and often is ignored, although it can be a sizable component of NPP. Insects and diseases can reduce productivity by as much as 25 to 50 percent in western forests (Stewart 1985; Stoszek 1973).

Table 1—Estimates of biomass and NPP for some western conifer forests. From Grier and others (1989). NPP includes above ground (A) and below ground (B)

Dominant tree species	Location	Age	Component	Biomass	NPP
		Years		*Mg/ha*	*Mg/ha/yr*
Abies amabilis mix	Washington	23	A	52	6.1
amabilis mix	Washington	23	A	77	17.9
amabilis mix	Washington	180	A	446	4.5
amabilis mix	Washington	180	A	583	16.7
lasiocarpa mix	Arizona	106	AB	357	8.6
Pinus ponderosa	Arizona	150	AB	162-250	4.9-5.7
monticola	Idaho	103	A	415-675	11.4-17.6
monticola	Idaho	103	A	488-794	13.1-20
monticola mix	Idaho	100-250	A	265-330	4.7-10
Pseudotsuga menziesii	Washington	22	AB	139	10.8
menziesii	Washington	36	A	172	13.8
menziesii	Washington	36	A	203	17.5
menziesii	Washington	40	A	249	9.9
menziesii	Washington	40	A	306	15.4
menziesii	Washington	73	AB	307	5.7
menziesii	Oregon	90-110	AB	661	12.7
menziesii	Oregon	150	AB	865	10.5
menziesii	Oregon	450	A	560	2.1
Tsuga heterophylla	Oregon	26	A	193	32.2
heterophylla	Oregon	26	A	231	37.7
heterophylla	Oregon	121	A	1,062	22.8

Climatic conditions favoring plant growth favor NPP. Thus, it is not surprising to see that the highest average rates of NPP occur in warm temperate and tropical forests, and the lowest are found in cold or arid regions (table 2). Lieth (1975), in analyzing NPP rates reported for all forest biomes, suggested that a productivity ceiling exists at a little over 30 Mg/ha/yr under the most favorable conditions of precipitation and temperature. Interestingly, dense young forests of *Tsuga heterophylla* (Raf.) Sarg. can achieve this level of NPP (table 1). Much, perhaps three-quarters of a site's total NPP, is directed below ground (Grier and others 1981), where a very high proportion soon becomes detritus from senescing fine roots and mycorrhizae. On the average, only 10 to 30 percent of total tree biomass is maintained below ground, but as much as three-quarters of ecosystem organic carbon is stored there and in the forest floor (table 2).

Just as with timber site quality, NPP has both an actual and potential component that varies with the same factors shaping wood production. Actual NPP increases steadily as a stand develops, and reaches a maximum rate with the advent of canopy closure, when the site has achieved its carrying capacity for foliage (Grier and others 1989). At this point, leaf mass stabilizes barring disturbance, and litterfall may represent a fairly constant proportion of stand NPP. Potential NPP depends then on stand age. However, at any given stage of forest development, potential NPP would be the total amount of dry mass produced per year when the site is fully stocked with vegetation. Grier and others (1989) provide an extensive discussion of problems and approaches in estimating above- and below-ground biomass and NPP. As we shall see, site productivity concepts have a direct bearing on how public forest lands are managed in the United States.

PUBLIC LAND LAW AND MONITORING REQUIREMENTS

The Multiple Use-Sustained Yield Act of 1960 (Sec. 4.[b]) binds the Forest Service to achieve and sustain outputs of various renewable resources without permanently impairing the productivity of the land (USDA Forest Service 1983). This mandate was reinforced in the National Environmental Policy Act of 1969 and the Forest and Rangeland Renewable Resources Planning Act of 1974. Directions were refined further by Sec. 6.(g)(3)(c) of the National Forest Management Act of 1976 (NFMA) which charges the Secretary of Agriculture with ensuring research and monitoring of the effects of each management system to protect the permanent productivity of the land (USDA Forest Service 1983).

In response to NFMA, an independent Committee of Scientists was appointed by the Secretary of Agriculture to help develop regulations for implementing the law. This led to a Code of Federal Regulations for Forest Planning which, among other stipulations, requires the Forest Service to monitor the effects of prescriptions, including "significant changes in land productivity" (Code of Federal Regulations 1985). The Chief of the Forest Service then directed each of the nine Forest Service administrative Regions to develop monitoring procedures for detecting significant changes in land productivity over a planning horizon (a rotation).

IMPLEMENTING THE REGULATIONS

The Forest Service's first task was to define the scope of its monitoring responsibility. "Land productivity" might encompass wildlife, watershed, fisheries, esthetic, and timber values. All of these are valid components of productivity, but they are not equally measurable. Some are intangible, subjective, or temporally unstable. Discussions with the Office of General Counsel and other parties helped form a more objective and usable definition of "land productivity" and "significant change" (USDA Forest Service 1987).

"Land productivity" was defined as a soil's capacity to support plant growth as reflected by some index of biomass accumulation. Although it may not be the broadest measure of productivity, plant growth is a useful index of ecosystem health. Losing a soil's plant growth capacity also means losing the site's capacity for sustaining other resource values. Further, a "significant change" in productivity was defined as the minimum level of reduced plant growth that is detectable using current technology.

Thus, the Forest Service is charged with protecting the inherent capacity of the soil to sustain plant growth, and to monitor the consequences of forest practices to the end that this capacity is not endangered. What, then, is "the inherent potential of the soil?" Assessing significant changes under operational field conditions is not simple, nor has

Table 2—Characteristics of organic carbon in vegetation above and below ground, the forest floor (FF), and mineral soil to a depth of 1 m in some major forest biomes. Modified from Powers and Van Cleve (1991), as compiled from several sources

Forest biome	Carbon above ground: NPP	Carbon above ground: Veg.	Carbon above ground: FF	Carbon above ground: Total	Carbon below ground: Veg.	Carbon below ground: Soil	Carbon below ground: Total	Total in ecosystem	Prop. in FF + b.g.
	Mg/ha/yr	*Mg/ha*							*Pct*
Semiboreal	2.4	76	29	105	24	159	183	288	74
Cool temperate	4.8	148	22	170	32	132	164	334	56
Warm temperate	10.6	84	10	94	20	96	116	210	60
Semiarid temperate	5.7	55	12	67	7	86	93	160	66
Subtropical	6.7	120	6	126	16	101	117	243	51
Tropical	10.1	157	4	161	15	107	122	283	44

the soil's potential been quantified. If current productivity is too variable, and if potential productivity is not easily assessed, could a substitute measure be used?

The Forest Service believes it is the soil. Along with climate, relief, and biology, soil sets the limits on productivity within a region through its control of nutrient, moisture, and air supplies to tree roots. "Soil productivity" is the term that describes this. If we knew the potential productivity inherent to each soil type, and the important factors that distinguish one soil from another, then the changes in those factors caused by management could be used to indicate changes in inherent, potential productivity. Thus, soil becomes a logical focus for monitoring. The Forest Service's Watershed and Air Management Staff has assumed responsibility for monitoring soil productivity. Monitoring strategy is based on three principles:

1. Management practices create soil disturbances.
2. Soil disturbances affect soil and site processes.
3. Soil and site processes control site productivity.

Monitoring soil and site processes directly is not feasible. Instead, monitoring focuses on measurable soil variables that reflect important site processes. For example:

Site process	Soil-quality monitoring variables
Soil erosion	Percentage soil cover or surface disturbance, soil bulk density, amount of soil loss, sediment production, presence of soil rills or pedestals, etc.
Water availability	Potential infiltration, saturated hydraulic conductivity, soil bulk density, puddling, plant moisture stress, soil water-holding capacity, etc.
Nutrient availability	Percentage soil cover, soil color and organic matter content, soil loss or displacement, etc.
Gas exchange	Soil bulk density or permeability, puddling, presence of mottles, water logging, etc.
Root growth and uptake	Soil structure, strength, or bulk density, water table depth, etc.

Soil scientists in each National Forest Region, with the approval of the Regional Forester, are directed to identify soil-quality monitoring variables judged to be important for their Region, to prepare Regional soil-quality standards, and to devise suitable methods for monitoring soil disturbances (USDA Forest Service 1987). This is aimed at ensuring the protection of soil productivity in the pursuit of land management objectives.

SOIL-QUALITY MONITORING STANDARDS

The principle behind the Forest Service approach is illustrated in figure 1A. For any given soil and site, a change in a key soil variable (for example, a loss in porosity) will lead ultimately to a change in potential productivity. If the soil variable is not closely linked with productivity, changes in the condition of that variable will have little or no bearing on productivity (potential productivity remains stable along the "line of no change" in fig. 1A). Obviously, a key feature is that the soil monitoring variable must have a close link with potential productivity.

The conceptual model in figure 1A is simplistic. It implies that potential productivity—the "dependent variable"—is a stable and known value. In reality, there is a belt of uncertainty surrounding a productivity estimate that is due to climatic vagaries and to limits in our knowledge. This uncertainty is shown as a shaded band about the line of no change in figure 1B. Uncertainty about the true value of potential productivity leads to uncertainty about how much change a soil can undergo before productivity is affected. Recognizing this, and based largely on collective judgment, the Forest Service estimates that a

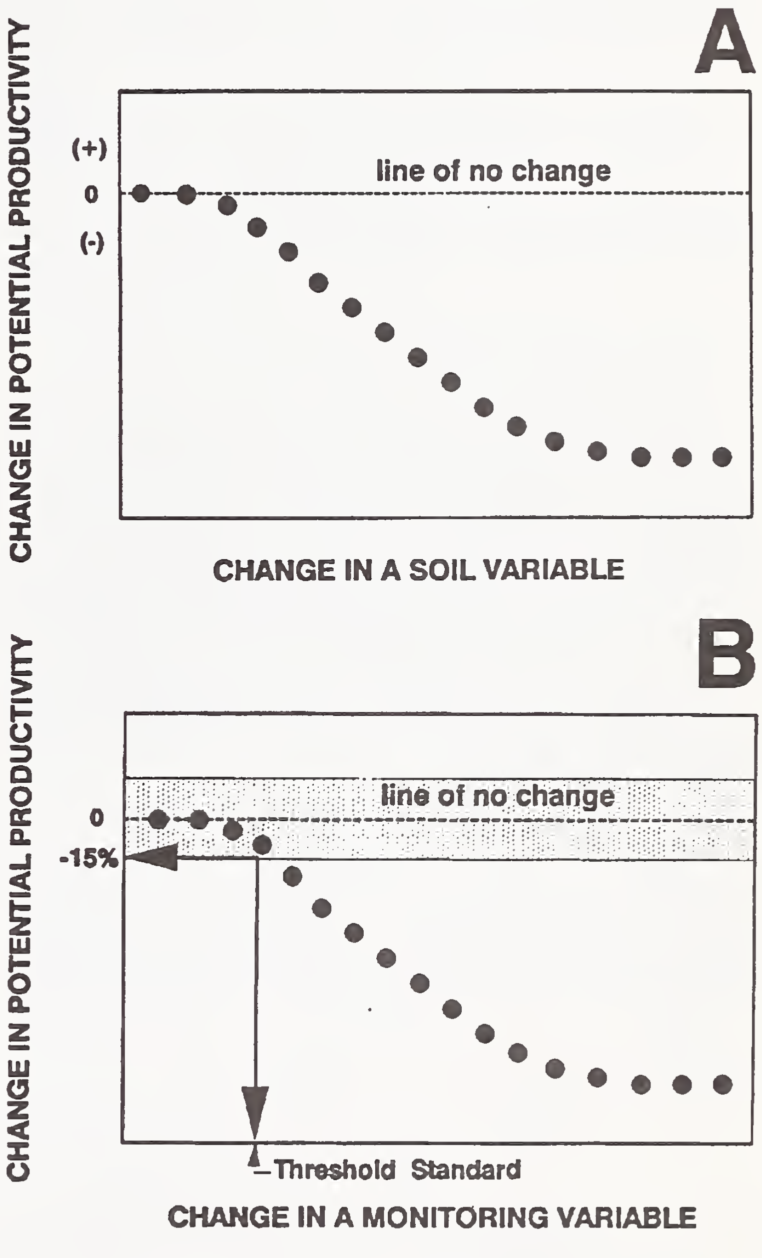


Figure 1—Hypothetical relationship between potential productivity and a key soil variable. A, as properties of a soil variable change, productivity declines from its potential before disturbance; B, in practice, a belt of uncertainty exists about the estimate of potential productivity for an undisturbed site. Threshold soil-quality monitoring standards are set at the level of soil change corresponding to a statistically detectable (15 percent) decline in potential productivity.

true productivity decline would need to be as great as 15 percent to be detectable by modern monitoring methods. Thus, soil-quality threshold standards are being set to detect a decline in potential productivity of at least 15 percent (fig. 1B). This does not mean that the Forest Service tolerates productivity declines of up to 15 percent, but merely that it recognizes problems with detection limits.

Soil-quality monitoring is seen as a three-stage process (USDA Forest Service 1987):

1. Implementation monitoring to ensure prescribed soil management practices are implemented as designed (a National Forest System [NFS] responsibility).
2. Effectiveness monitoring to determine the effectiveness of prescribed soil management practices (a NFS responsibility).
3. Validation monitoring to determine whether monitoring standards and guidelines are appropriate to maintain soil productivity (a Research responsibility).

What is the technical basis for soil-quality monitoring standards? Often, they are based on the collective judgment of professionals because the subject has not been tackled from a direct, scientifically rigorous perspective. But is there sound evidence that soil property changes are associated with substantial declines in productivity?

HAS PRODUCTIVITY DECLINED?

Effects of massive slope failure and loss of the soil mantle on long-term productivity are so obvious that they need not be documented here. But do less dramatic activities also degrade? Evidence from the United States tends to be confounded, or short term and inconclusive (Powers and others 1990). The best clues come mainly from abroad where forestry has been practiced longer, records are more complete, or multiple short rotations have occurred. World findings have been reviewed recently by Powers and others (1990), and the main points are summarized here.

Removal of Biomass

Utilization standards affect rates of biomass and nutrient removal, and nutritional aspects have drawn most of the research attention. Generally, one-tenth of total ecosystem nitrogen and a smaller fraction of ecosystem phosphorus is contained in the biomass of young, mature forests (table 3). Of this, at least half of the nitrogen and phosphorus is contained in foliage and branches. As a rule, a 1 percent increase in biomass removal means a 3 percent increase in nutrient removal (Switzer and others 1981). Direct evidence of productivity decline from biomass removal is rare. Sterba (1988), in a thinning study in Austria, found that residual trees had 12 percent greater growth if felled trees remained on the site than if they had been removed. However, the reason for this remains speculative.

Organic matter and nutrient losses in conventional timber harvests and rotations seem unlikely to affect potential productivity on most sites (Wells and Jorgensen 1979). However, shorter rotations deplete nutrients much more rapidly than do long rotations. Switzer and others (1981) estimated that with whole-tree harvesting (the removal of all aboveground biomass), three 20-year rotations in loblolly pine would remove three-quarters more organic matter and twice the nitrogen as one 60-year rotation. Even so, this amounted to less than 10 percent of the nitrogen in the ecosystem. To date, assessments of the impacts of increased utilization or shortened rotations are based mainly on model projections and not empirical evidence. They are predicated on many assumptions lacking rigorous proof.

Loss of Organic Matter

In South Australia, Keeves (1966) showed that radiata pine planted on sandy soils grew much more poorly in the second rotation than in the first. Squire and others (1985) demonstrated that the decline was triggered by the burning of logging slash, which led to nutrient and moisture stress. The effect was evident by the time of crown closure. They showed that the growth of second rotations could exceed that of first rotations if logging residues were retained. Findings suggest that organic matter retention is particularly important on infertile, droughty sites.

Organic matter in the forest floor (surface residues of plants and animals that are not yet soil) has singular significance in respect to productivity. Although its mass is only a small fraction of that in the standing forest (table 2), its nutrient content often equals or exceeds the combined total for the trees and understory vegetation (table 3). Substantial losses of forest floor material have degraded some sites. Litter raking—the regular removal of freshly

Table 3—Ranges in total nitrogen (N) and phosphorus (P) contents reported for ecosystem components of young, mature true fir, pine, and Douglas-fir forests in North America. Modified from Kimmins and others (1985)

Ecosystem component	True fir		Pine		Douglas-fir	
	N	P	N	P	N	P
	kg/ha					
Trees						
Above ground	80 - 686	12 - 83	180 - 556	12 - 31	84 - 728	18 - 112
Below ground	24 - 72	4 - 12	12 - 117	2 - 21	30 - 90	5 - 18
Understory	2 - 50	[1]t - 14	1 - 54	t - 5	5 - 66	1 - 9
Forest floor	666 - 2,300	9 - 103	80 - 1,240	9 - 103	110 - 1,249	19 - 115
Soil to 1-m	5,237 - 14,000	3,212 - 6,317	1,753 - 5,554	146 - 4,457	1,170 - 15,400	3,878 - 3,900

[1]t = trace.

fallen conifer needles—was practiced in central European conifer forests for centuries to obtain bedding straw for farm animals. Wiedemann (1935) found that several decades of litter raking in Scots pine plantations on sandy soils in eastern Germany led to higher soil densities and to growth declines of nearly two site classes. Soils also were lower in fertility (Baule and Fricker 1970). Similar results were found for radiata pine in New Zealand, where 26 years of litter raking led to appreciable declines in soil nutrients, an increase in bulk density, and about a 12 percent decline in growth (Dyck and Skinner 1990).

Loss of Soil Porosity

Sands (1983), working with radiata pine in Australia, found sizable increases in soil density on sites converted from pasture to pine plantations. The effect increased by the second rotation because of multiple passes of machinery, and approached a growth-limiting density at depth. In the United States, Froehlich and others (1986) and Helms and others (1986) reported that trees growing on compacted soils contained only about one-fifth the volume of trees growing on less compacted soils nearby. Growth losses depend on the degree of compaction. The greater the proportional increase in soil density, the greater the productivity loss (Froehlich and McNabb 1984).

Freezing and thawing promotes natural recovery. But for coarse-textured soils on warmer sites, effects largely are irreversible (Sands 1983). And on finer textured soils, natural recovery may take decades (Hatchell and others 1970). Sands (1983) concluded that desirable soil physical properties depended largely on the maintenance of soil organic matter. In time, soils will compact of their own weight without further disturbance if organic matter is lost appreciably from the profile (Sands 1983).

Loss of Topsoil

Although soils often contain vast amounts of nutrients (table 3), relatively small losses of surface soil can affect productivity. This is because many nutrients, such as nitrogen, are concentrated in organic matter. Organic matter, in turn, is not distributed evenly throughout the soil, but is concentrated near the surface and decreases rapidly with depth (Powers 1989). In New Zealand's pumice region, displacing logging debris and a thin layer of topsoil into windrows during site preparation produced nutrient deficiencies and led to a 30 percent loss in volume growth by mid rotation (Dyck and Beets 1987). Similar results were reported in the United States for loblolly pine in North Carolina (Fox and others 1989), for ponderosa pine in California (Powers and others 1988), and for Douglas-fir in the Pacific Northwest (Minore 1986). Tew and others (1986), examining practices in the Piedmont region of the southeastern United States, estimated that piling logging residues into windrows removed two to three times more nitrogen and phosphorus than whole-tree harvesting.

Topsoil also is lost through erosion if surface organic matter is removed appreciably or if surface soils are compacted. Removing the forest floor exposes a soil to rainfall, particle displacement, and sealing of surface pores. Depending on slope steepness, this can lead to erosion rates 40 to 1,200 percent greater than those caused simply by logging (Megahan 1987). Compaction or topsoil displacement exposes a soil surface with a lessened capacity to absorb rainfall, and leads to greater runoff and erosion. Although erosion following timber harvest and site preparation can be equivalent to rates reported for agriculture (Neary and others 1984), the period of accelerated loss tends to be brief as the site revegetates. Effects of surface erosion on long-term productivity are not well known and can only be inferred.

Summarizing the Evidence

Substantive losses of surface organic matter and soil porosity have led to documented declines in productivity. Organic matter and porosity influence productivity through their link with more fundamental processes, as indicated by the conceptual model in figure 2. But beyond conceptual models, we lack a specific understanding of what a given change in porosity or organic matter means in terms of its long-term effect on productivity. As Miller and Hazard (1987) have said, uncertainty and skepticism will persist until we establish and maintain studies that help us document and understand the long-term effects of forest practices on productivity.

Although soil-quality monitoring standards now in use tend to focus on changes in porosity and organic matter, they are based mainly on "best professional judgment" from extrapolation of anecdotal studies or from general observations that are subject to various kinds of bias. Definitive calibrations such as depicted in figure 1 simply do not exist. Because of this, many standards may face legal challenge as to whether they are too restrictive, or not restrictive enough. The effectiveness of current standards must be validated. If standards are found wanting, they must be improved.

Work is needed at both the fundamental and applied level to quantify the effects of soil disturbance on potential productivity—not only in a timber management sense, but in a more fundamental sense as well. We know little about the inherent carrying capacity of forest sites for producing vegetation, or about its relationship to key soil variables. Yet, this relationship must be known if we hope to produce accurate calibrations (fig. 1A) and effective soil-quality monitoring standards (fig. 1B). Such calibrations must be developed if the Forest Service is to meet its legal and ethical responsibility. Clearly, Forest Service Research has the mandate for conducting such work. But the scope and scale of the problem are far too vast for Forest Service Research to tackle alone. There must be a partnership.

A COOPERATIVE NATIONAL STUDY

In December 1986, at the annual meeting of the Soil Science Society of America in New Orleans, an informal session was held for Forest Service attendees. There, P. E. Avers—National Soils Group Leader from the Washington Office—outlined the problem of soil-quality monitoring facing NFS, and asked for assistance from Research. Further discussions were held between Avers, D. H. Alban, and R. F. Powers. The following June, Avers and four scientists from the North Central, Pacific Northwest, Pacific

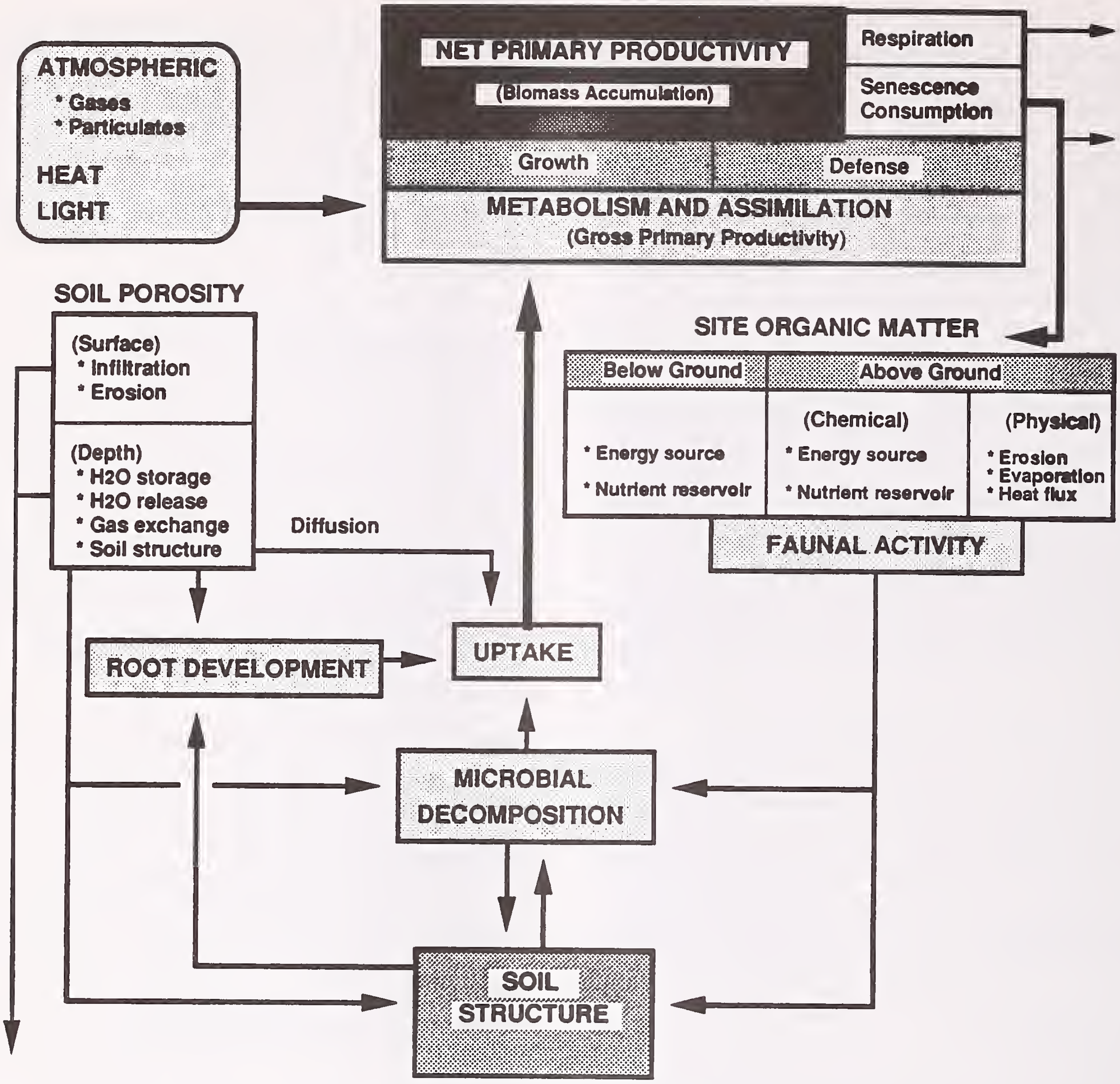


Figure 2—Conceptual model of the roles of soil porosity and site organic matter in regulating the processes controlling site productivity.

Southwest, and Southern Forest Experiment Stations met in a California field setting to review the scope of the problem and discuss possible solutions. This was followed by a March 1988 session in St. Louis to plan a formal strategy for Research and NFS collaboration. Attendance was expanded to include a scientist from the Southeastern Station and one representative each from Timber Management, Timber (now Forest) Management Research, and Forest Environment Research staffs of the Washington Office.

These meetings led to the creation of two major documents. The first was an extensive, critical literature review concerning the long-term impacts of forest practices on potential productivity. Part of this was a proposal for a new national research effort. The paper was presented before an international gathering of scientists in July 1988, and was published in the peer-reviewed proceedings of the conference (Powers and others 1990). The second document was a plan prepared in October 1988 by Research and NFS personnel outlining a cooperative national Forest Service study of the impacts of management practices on long-term soil productivity (LTSP). The plan was presented to the Chief's staff and received extensive technical review both in the United States and abroad. In 1989 it was approved by the Washington Office (Powers and others 1989), and the plan now serves as a template for more detailed regional studies.

Objectives

Cooperation is under way between Forest Service Research and the National Forest System with the objectives of (1) quantifying the effects of soil disturbance from

management activities on soil productivity with the aim of establishing site-specific calibrations such as are hypothesized in figure 1B; (2) validating standards and techniques for soil-quality monitoring; and (3) understanding the fundamental relationships between soil properties, long-term productivity, and forest management practices (Powers and others 1989). These objectives can be stated as complementary "Research" and "Development" topics.

Research	**Development**
How does soil disturbance affect:	*Facilitate soil monitoring efforts by:*
• Carbon allocation.	• Calibrating changes in soil properties against (i) stand productivity; and and (ii) total productivity.
• Water use.	
• Nutrient use.	• Evaluating/developing practicable field monitoring methods.
• Other processes.	
• Resistance to pests.	• Developing means for extending results to a broad array of sites.
• Fundamental productivity.	

A fundamental purpose of the study is to develop understanding of the joint role of soil porosity and site organic matter in their effect on site processes controlling productivity. An applied purpose will be to validate soil-quality standards and monitoring methods used by NFS, and to develop new procedures as needed. Highlights described later are condensed from Powers and others (1989).

Coordination

Three levels of coordination are described to ensure the achievement of national and regional objectives in the national LTSP program.

National Oversight—National coordination and review for LTSP is provided by Forest Management Research, Forest Environment Research, Timber Management, and Watershed and Air Management Washington Office staffs. Coordinating activities include:

1. Developing priorities and funding opportunities.
2. Reviewing Station/Region implementation plans.
3. Highlighting accomplishments.

National Technical and Regional Steering Committees are responsible for establishing and conducting the LTSP studies.

National Technical Committee—The National Technical Committee for LTSP consists of Principal Investigators and Regional Soil Scientists who presently are, or who expect to be, involved in implementing and maintaining studies and interpreting study results. This committee, chaired by a Principal Investigator appointed by the National Oversight Committee, will have five main responsibilities:

1. Reviewing study proposals to ensure that scientific methods are consistent and appropriate to meet program objectives.
2. Establishing a national data base of research results.
3. Informing the Washington Office of progress, needs, opportunities, and substantive findings.
4. Coordinating and preparing results for publication.
5. Reviewing, evaluating, and incorporating modifications to the proposals.

Regional Steering Committee—The Regional LTSP Steering Committee consisting of the Experiment Station Principal Investigator, Regional Soil Scientist, and Regional Silviculturist initiates and encourages cooperative and collaborative ties with National Forests, Ranger Districts, and other researchers. This committee identifies individual study sites, prepares specific study plans based on this general plan, and implements the studies. The Regional Committee shares responsibility with National Forests and Ranger Districts for ensuring public awareness of the program. Departures from procedures described in the study plan must be approved by both the National Technical Committee and the National Oversight Committee. Study sites should be identified in the Monitoring Plan for each National Forest.

Affiliations—Prospective collaborators from a dozen U.S. universities have formed an ad hoc committee to explore ways to secure funding for joint LTSP research with Forest Service scientists. International collaboration also is under way through British Columbia's Ministry of Forests and New Zealand's Forest Research Institute for parallel LTSP studies using our protocol.

Experimental Protocol

Treatments—A broad range of manipulations of soil porosity and site organic matter are applied on benchmark soils within the major commercial forest types of the United States. Work began in 1989 in the Pacific Southwest and Southern Regions on lands expected to be managed intensively in the decades ahead. In 1990 and 1991 the LTSP program expanded to the Northern and Eastern Regions (fig. 3), and it continues to expand both domestically and abroad.

Figure 3—The commercial forest region of the conterminous United States showing national LTSP installations. Circles: sites installed or undergoing installation. Squares: sites proposed for installation in 1992. Some symbols represent more than one installation.

Within each Region and forest type, and across a broad range of productivity classes, about a dozen timbered sites will be selected for treatment. Sites will be characterized before treatment according to a standard protocol (Powers and others 1989). A core series of organic matter removal and soil compaction treatments will be assigned randomly to 0.4-ha treatment plots. All possible combinations of the following main effects will produce nine core treatments to be applied at each site.

Main effect	Treatment level	
Organic matter removal	OM0	Boles removed, only
	OM1	Boles and crowns removed (whole-tree harvesting)
	OM2	Boles, crowns, understory, and forest floor removed (all aboveground biomass)
Soil compaction	C0	No compaction
	C1	Intermediate compaction (halfway between C0 and C2)
	C2	Compaction to about 80 percent of the difference between hypothetical growth-limiting bulk density (Daddow and Warrington 1983) and bulk density existing before treatment at 10-20 cm.

This cluster of nine treatment plots will cover the range of site organic matter and soil porosity changes apt to occur under present or future forest management. Other treatments—such as topsoil removal, conventional harvest and site preparation techniques, or ameliorative practices—may be installed at the prerogative of the Regional Steering Committee, provided that they do not confound or alter the core cluster of nine treatments. Work has begun on about a dozen U.S. sites as of this writing, and another dozen should be on line by 1992 (fig. 3).

Generally, each 0.4-ha treatment plot will be planted with seedlings of the appropriate timber type and a mixture of the best available genetic stock. Natural regeneration (seeding or sprouting) may be substituted where appropriate. The aim is to favor superior growth without narrowing genetic diversity. Each treatment plot will be split in half, creating two 0.2-ha subplots of about 340 trees each (fig. 4), with a measurement plot established from the fourth row of trees inward in each subplot. After establishment, one subplot will be kept weed free. In the other, regional vegetation will be encouraged to grow with the trees—the aim being to promote complete vegetative recovery as rapidly as site conditions permit so that the site approaches full carrying capacity for vegetative growth. This split-plot arrangement creates a means for side-by-side comparisons of (1) stand productivity vs. total vegetative productivity, and (2) the effect of competing vegetation on tree growth.

Measurements—Soil properties capable of being monitored operationally will be measured periodically and correlated with vegetative growth. Because all treatment plots

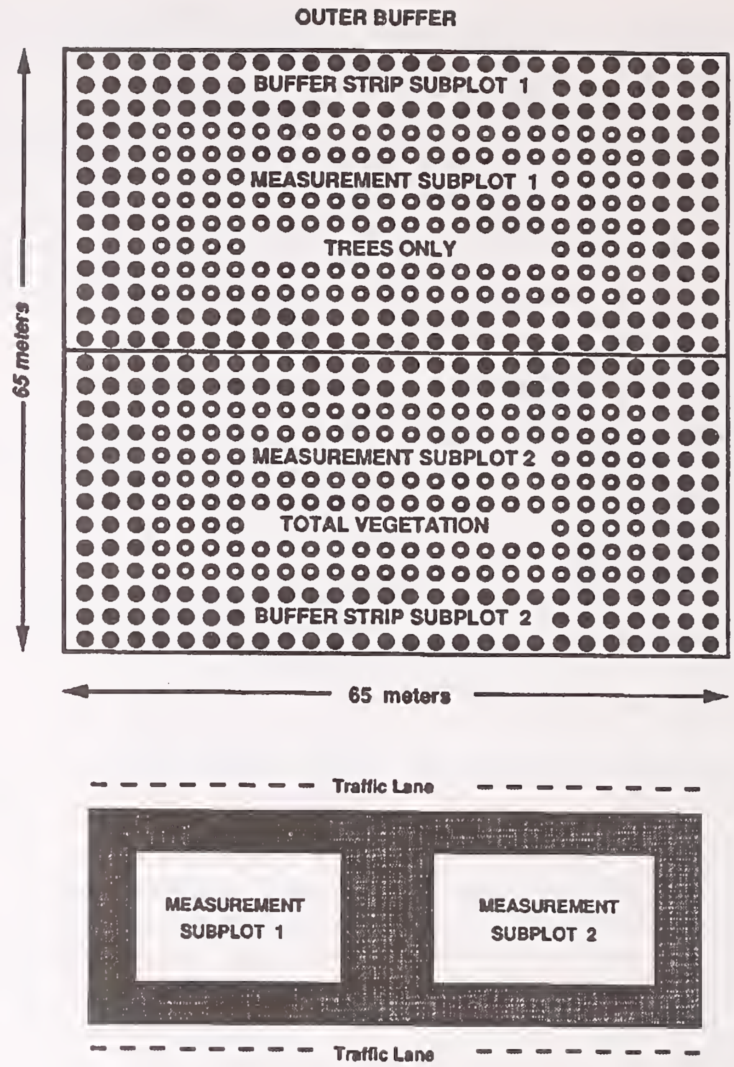


Figure 4—Possible layouts for 0.4-ha treatment plots and 0.2-ha subplots. Tree locations are indicated by circles. Each measurement subplot contains three rows of buffer trees (dark circles) along its outer border. The square plot (upper) would be appropriate for full-suspension aerial harvesting systems. The rectangular plot (lower) is possible where materials can be removed by a loader positioned in a traffic lane just outside the plot.

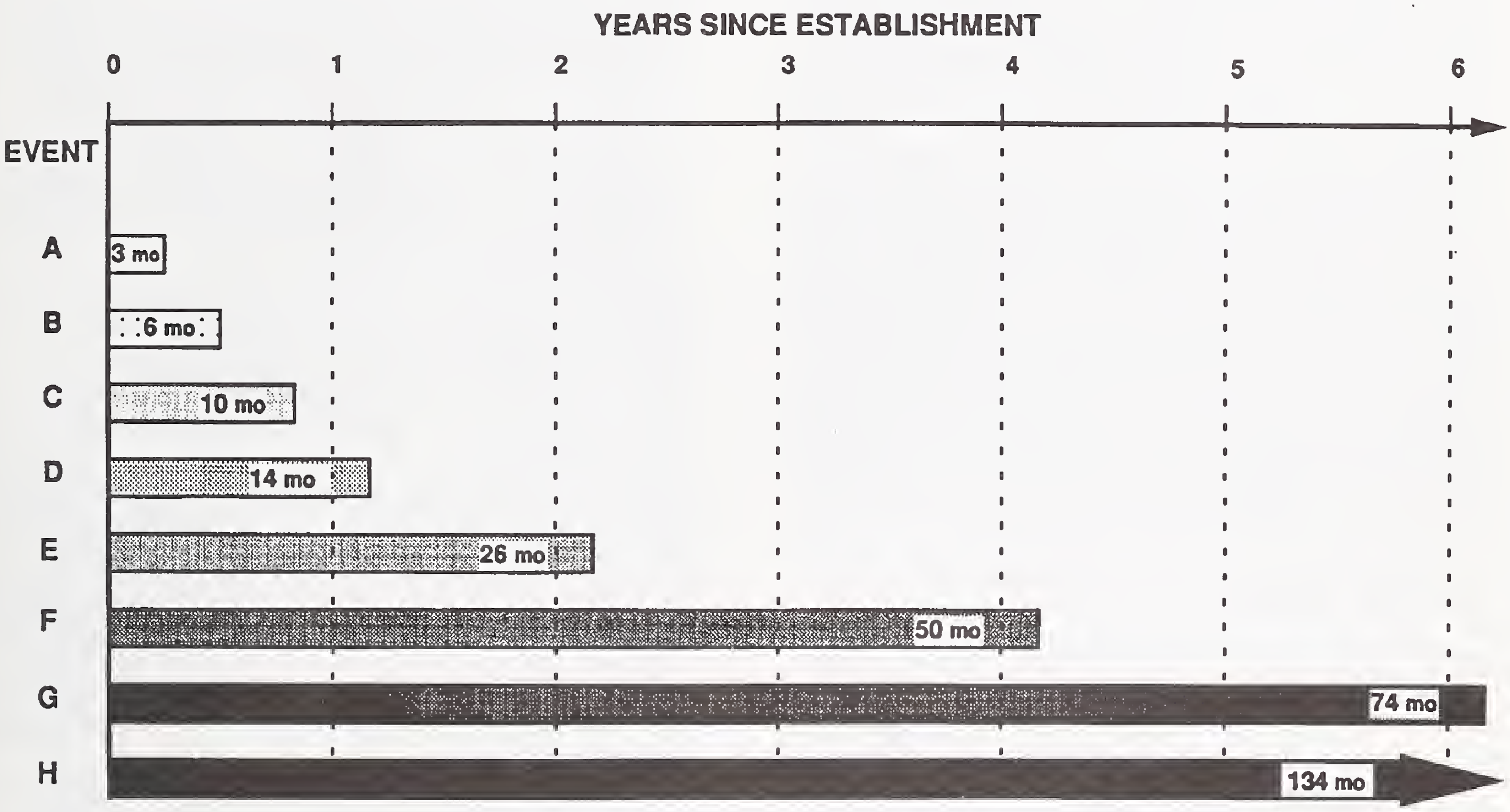
are adjacent in an installation, comparative rates of vegetative growth will provide a precise measure of differences in site productivity. Wood and total tree biomass production in "trees only" subplots provide the basis for traditional "timber site quality" measures as affected by soil disturbance, with "OM0,C0" treatments providing a baseline control. Also, the "total vegetation" subplots provide a more comprehensive measure of "total site productivity" as reflected in total NPP. Productivity measured near the point of crown closure will provide a precise estimate of maximum potential productivity. And throughout the study, comparing each treatment against the OM0,C0 control will be an excellent means for judging whether soil changes have affected the potential productivity of the land. Measurements are:

Pre- and posttreatment measurements dealing with—		Postplanting measurements dealing with—
Organic matter	Porosity	Productivity
Mass by component	Bulk density	Stocking
• Logging slash	Soil strength	Height
• Forest floor	Infiltration	Diameter
• Mineral soil	Moisture release	Damage
Nutrients by component	Saturated hydraulic conductivity	Aboveground biomass in:
• Logging slash	Erosion	• Trees (by component)
• Forest floor	Soil temperature	• Other vegetation (by component)
• Mineral soil		Carbon partitioning
Decomposition and mineralization		Species diversity

A standard weather station and datalogger will be installed at each site to monitor air and soil temperature, wind speed and direction, relative humidity, total and photosynthetically active solar radiation, precipitation, and evapotranspiration. Stations will be compatible with others installed throughout the country, and will add to our monitoring base for detecting climatic change and its possible impact on the productivity of the land.

Responsibilities and Implementation

Costs, estimated at about $74,000 per installation for the first 5 years, will be shared by NFS and Research. In general, NFS is responsible for establishing and maintaining the sites. Research is responsible for data collection, analysis, and publication. Study plans prepared jointly by Stations and Regions will detail the responsibilities. Contacts and field visits with research and practicing entomologists, pathologists, and wildlife specialists should be frequent. Once funded, approximately 14 months will be the minimum time needed to locate and install a series of study sites within a timber type (fig. 5). The first strong indication of long-term treatment effects can be expected in 5 to 10 years after planting on average sites.



EVENT

A: Develop regional study plan with approval.
B: Identify candidate study sites.
C: Select study sites, preliminary sampling, develop contracts and special provisions.
D: Harvest, install treatments, plant, collect initial measurements.
E: First main post-treatment measurements.
F: Second main post-treatment measurements.
G: Third main post-treatment measurements. First biomass measurements.
H: Fourth main post-treatment measurements. Second biomass measurements. (Probably the first valid indication of long-term productivity trends).

Figure 5—Sequence and timeline for locating, installing, and measuring study plots. Trees would be thinned according to best management practices and the stand characteristics of that treatment. Sampling would continue throughout the rotation.

SUMMARY

Our ability to maintain a site's productive capacity faces increasing challenge through public review of Forest Land Management Plans and timber sales. In response to NFMA, Forest Service Regions are developing threshold soil-quality standards for detecting declines in potential soil productivity. Such standards are based on best available information. Often, this amounts to professional judgment, because research has not addressed the problem squarely. Professional judgments will be subject to repeated challenge from many sectors. Results from this cooperative study will provide credible responses to many of these challenges, and will address related research needs across the Nation. An immediate benefit is a clear show of good faith by the Forest Service to adhere to the spirit of NFMA and to tackle the problem aggressively.

This national network of study sites will provide the scientific basis for validating soil-quality standards established by NFS, and creates a research opportunity of unusual scope and significance. Initially, existing standards for monitoring soil quality will be compared with interim findings and can be adjusted to reflect the most recent research results. With time, more substantive results will be available for future planning. All Regions and Stations are encouraged to participate to ensure the success of this joint program.

The experimental design provides researchers with the framework for comparing stand production with more fundamental measures of productivity. Because a gamut of stress conditions is imposed deliberately on vegetation, pest and disease interactions are likely, and multidisciplinary collaboration will be encouraged. Basic models of soil and growth processes can be integrated with site and climatic data to extrapolate findings to a broad array of sites, and to project the possible impacts of changing climate on future productivity. The work fosters close cooperation between Research and NFS, and opens doors for important work with university and industry colleagues.

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INFLUENCES OF VOLCANIC ASH AND PUMICE DEPOSITION ON PRODUCTIVITY OF WESTERN INTERIOR FOREST SOILS

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ABSTRACT

This paper is a synthesis of published research pertinent to soils developed from Mount Mazama tephra deposits in the Pacific Northwest. The research discussed includes studies of physical and chemical properties, of changes in these properties imposed by forest harvest, of relations between altered properties and tree growth, and of vegetation responses to fertilization.

INTRODUCTION

Recent and historical volcanic activity has deposited ash and pumice on portions of the western United States and affected the productivity of some forested areas. Our paper focuses primarily on soils developing in ejecta from Mount Mazama in Oregon. We have utilized research findings pertinent to these soils as examples of what may exist in soils developed from other pumice and ash sources, both local and more general. We also recognize that considerable variablility is likely in these widely occurring parent materials.

Our objectives are to overview productivity relationships of these soils to physical and chemical properties, to harvest-induced changes in these properties, and to soil fertility management in tests of nutrient additions from fertilizers.

McCool (1914) made early observations of physical and chemical properties of pumice soils in central Oregon. He found these soils had high porosity, high water-holding capacity, and rapid infiltration rates. According to Fryxell (1965) the eruption of Mount Mazama occurred some 6,600 years ago and affected about 900,000 km^2 to the north and east of the source, now known as Crater Lake, in Oregon. Particle size and thickness generally decline with increasing distance from the source. Particles near the source are highly porous, gravelly pumice but are dominantly silt sized at the extremities of the deposition. The ejecta near the source formed a pumice-mantled plateau of about 50,000 km^2 in central Oregon. At greater distance from the source, the finer deposition also imposed important changes, but the soils are distinctly different from pumice soils (Geist and Strickler 1978).

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SOIL PHYSICAL AND CHEMICAL PROPERTIES

About 64 km (40 miles) from the source a typical soil profile has a sandy loam A1 horizon about 5 cm thick, a 10- to 30-cm-thick loamy sand AC, a 70-cm gravelly loamy sand C1, and a 50+-cm loamy sand C2 over a IIB and/or IIC buried horizon. This soil and closely associated soils of the pumice-mantled plateau have recently been classified as Xeric Vitricryands.

The C1 horizon contains coarser textured material of special interest, because the individual pumice particles are bridged together resembling a three-dimensional jigsaw puzzle. The bridging restricts root proliferation (Cochran 1971; Hermann and Petersen 1969). Sinker roots of ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) usually penetrate through the C1 horizon, often through old root channels and rodent burrows. Roots of lodgepole pine (*Pinus contorta* Dougl. ex Loud.) are mainly restricted to the A1 and AC horizons. Because limited root penetration of the C1 and C2 horizons severely alters the positional availability of both water and nutrients, productivity is lowered where these horizons are present. The restricted rooting also makes trees more vulnerable to windthrow.

At greater distance from the source the mantle of pumice is thinner and the particle size smaller. The AC horizon becomes thicker, the C1 thins or disappears, and restrictions on root growth lessen or become nonexistent. At these distances more mixing of weathered and unweathered material has apparently resulted from windthrow, animal burrowing, and redistribution by water and wind action. Because of this alteration, the C1 horizon is mixed in varying degrees with finer material, so particle bridging is much reduced, and productivity is generally higher than where mixing has not occurred.

An older, preeruption, buried soil occurs below the Mazama pumice layer at depths varying from 50 cm to over 3 m. Hermann and Petersen (1969) found sharp increases in height growth of young ponderosa pine (8-20 feet tall) where root systems penetrated the C1 and C2 horizons and reached the buried soil profile.

Thermal properties of these soils combined with climatic conditions of the area result in frequent radiation frost during the growing season. Frost heaving can also be a

severe problem in limiting natural regeneration and can completely eliminate a fall planting.

Radiation frost damage is a general problem in the pumice zone because of the high elevation, the dry air mass, the predominance of clear night skies, and the thermal properties of the soils. Pumice soils have thermal properties that more closely resemble peat soils than common mineral soils. Thermal conductivities, thermal diffusivities, and thermal contact coefficients are low. Therefore, surface temperatures are extremely warm in the day and quite cold at night (Cochran 1975b). Damage from radiation frost is accentuated in basins and flats where cold air accumulates. Lodgepole pine is better suited to these areas than ponderosa pine because of greater tolerance of germinating lodgepole seedlings to temperatures below –7.8 °C (Cochran and Bertnsen 1973), and because the female cones of lodgepole pine are less susceptible to damage at low temperatures (Sorensen and Miles 1974). Ponderosa pine is more predominant on raised areas or uplands (Dyrness and Youngberg 1966; Youngberg and Dyrness 1959), and the scattered ponderosa pine in lodgepole areas on flats and basins apparently becomes established only beneath the shelter of larger lodgepole pine. Young ponderosa pines in basins and flats require sheltering until they reach about 2 m height. Planting of ponderosa pine in these areas without overstory protection results in stunted plants (Cochran 1984). Natural regeneration difficulties will be extreme after clearcutting, and even lodgepole pine may require planting if regeneration is desired within 5 years. Pumice deserts created by cutting or fire are hard to artificially regenerate, and natural regeneration could take many years. An ecologically more compatible silvicultural strategy could be to maintain lodgepole pine and use shelterwoods without complete slash cleanup to help regenerate new stands. Such an approach will perpetuate the existence of an onsite seed source of lodgepole pine for ensuing crops, while providing some protection to scattered ponderosa pine which may occur (Cochran 1975b). Conversions to pure ponderosa pine should be avoided.

Frost heaving is another problem accentuated by the soils and climate of the pumice-mantled plateau. Frost heaving begins when soils are moist to the surface and day-to-night temperatures fluctuate somewhat above and below freezing. Water in the surface layer freezes rapidly after sunset, creating an upward gradient in soil water potential. Water continues to move upward and freeze, and the pore space in a thin soil layer at the surface becomes filled with ice. As more water continues to move up from the unfrozen layer and freezes at the ice-water interface, ice crystals elongate downward and separate the frozen and unfrozen soil layers. Plants with root collars encased in the frozen layer can be completely or partially heaved. Fall planting will be completely heaved if not covered immediately by snow, which persists until spring. Therefore, only spring planting is recommended. To protect natural seedlings from frost heaving some uncrushed logging slash should be retained on the soil surface. Shelterwood cutting offers little, if any, protection from frost heaving problems, but does reduce radiation frost problems (Cochran 1975b).

At 400 km (250 miles) from the source, finer particle deposition produced ash soils that lack the severe thermal problems inherent in central Oregon pumice soils, and forest associations differ as well. In eastern Oregon and Washington, western and northern Idaho, western Montana, and southern British Columbia and Alberta, volcanic ash soils are associated with several forest types including: lodgepole pine, western larch (*Larix occidentalis* Nutt.), Douglas-fir (*Pseudotsuga menziesii* var. *glauca* [Biessn.] Franco), grand fir (*Abies grandis* [Dougl. ex D. Don] Lindl.), mixed conifer, spruce-fir, and others. Mixed conifer may include grand fir, interior Douglas-fir, western larch, subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.), western white pine (*Pinus monticola* Dougl. ex D. Don), and Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) in eastern Oregon and Washington. Farther east and north, western hemlock (*Tsuga heterophylla* [Raf.] Sarg.) and western redcedar (*Thuja plicata* Donn ex D. Don) may also occur in the mixed conifer type. Ash soils are usually associated with the most productive forest sites.

Ash profiles in eastern Oregon commonly have 50-60 cm of light-colored, weakly structured, silt-loam textured ash overlying 20+ cm of a darker colored buried soil of residuum or colluvium commonly from basalt or andesite. The buried material is silty clay loam to clay loam in texture. No severe rooting restrictions are naturally present. These ash soils and related andic intergrades include Vitrandepts, Cryandepts, and Eutroboralfs. A common profile sequence has 5-10 cm of A horizon, 30-40 cm AC, 10-20 cm C, overlying 10-20 cm of IIB and IIC. Colors in the ash overburden lighten markedly with drying. In northern Idaho and northwestern Montana ash overlies granitic materials, including those weathered from intrusive rock or compacted glacial till, the latter being restrictive to rooting. Some of these soils have been reported as Dystrochrepts, Dystrandepts, and Fragiochrepts (Niehoff 1985; Nielson-Gerhardt 1986).

Some of the distinctive properties of volcanic ash and pumice soils are evident when compared to basalt-derived soils (table 1). The 0- to 30-cm depth of ash and pumice soils has lower bulk density, is lower in clay content, has fewer coarse fragments, and has higher potentially available soil-water storage. A high water-holding capacity, roughly twice that of an ordinary coarse-textured soil,

Table 1—Generalized comparisons of physical properties of ash-, pumice-, and basalt-derived soils, 0-30 cm depth, (adapted from Cochran 1966; Geist and Strickler 1978; Youngberg and Dyrness 1964)

Property	Ash Eastern Oregon	Pumice Central Oregon	Basalt Eastern Oregon
Bulk density, Mg/m^3	0.7	0.6	0.9
Porosity, percent by volume	73	77	65
Clay, percent by weight	10	8	20
Coarse fragments, percent by volume	4	25	30
Available water, percent by volume	30	30	13

is one of the unique features McCool (1914) found in early investigations of pumice soils. The clays of ash and pumice soils are dominantly amorphous, but they do contain minor amounts of layer silicates (Chichester and others 1969; Fosberg and others 1979; Harward and Youngberg 1969).

Ash and pumice, which have low volume weight and relatively few profile coarse fragments, can easily be moved by heavy equipment. Consequently, their vulnerability to mechanical displacement is high, especially when soil moisture levels are low. Even though they have naturally low bulk densities, ash and pumice soils will compact to growth-affecting levels, as is discussed further in subsequent sections. Under natural conditions ash and pumice soils generally have moderate to high infiltration rates and are well drained (McCool 1914).

Desorption studies of soil moisture content at various soil moisture tensions show ash materials have higher water content at lower tensions, similar water content at higher tensions, and yield more water in the lower tension range, as compared to basalt-derived materials (fig. 1). Given fully charged profiles, volcanic ash can supply more readily available water to plants than can basalt-derived soils (Geist and Strickler 1978). The relation between moisture content and moisture tension for pumice soils is very similar to that of ash soils, although the relation does vary among pumice horizons because of different amounts of vesicular pumice particles that exceed 2 mm in diameter. These coarser materials add to the water-holding capacity of the profile, but they exhibit slow unsaturated water movement because of disrupted pore continuity between and within pumice particles (Cochran 1966; Youngberg and Dyrness 1964).

Chemical properties will be addressed in relation to vegetation to illustrate some associations with chemical attributes. In central Oregon, both ponderosa and lodgepole pines are associated with pumice soils. In eastern Oregon, ponderosa pine is more often associated with

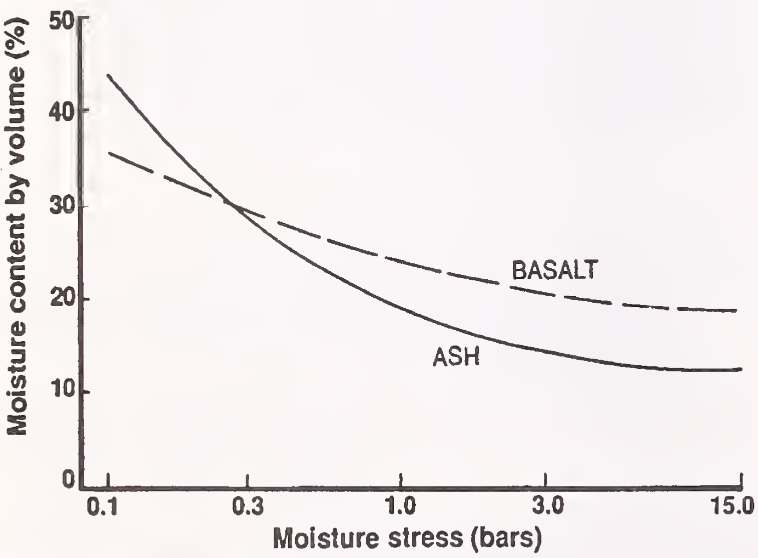

Figure 1—Generalized relationships of volumetric moisture content to soil moisture stress in volcanic ash- and basalt-derived soils (adapted from Cochran 1966; Geist and Strickler 1978; Youngberg and Dyrness 1964).

basalt- and andesite-derived soils; while lodgepole pine, western larch, interior Douglas-fir, mixed conifer, and spruce-fir types are associated with volcanic ash soils (Geist and Strickler 1978).

As in most soils, organic matter, total N, and extractable P are higher in the surface layers of ash and pumice soils and decline with depth (table 2). This orientation appears to be exaggerated, though in varying degree, in pumice soils. Extractable cations are less surface oriented, and in fact, commonly occur in higher concentrations, deeper in the profile. Youngberg and Dyrness (1964) found that profile quantities (kg/ha) of extractable cations could increase 1.5-2 times where thicker pumice C horizons occurred. Geist and Strickler (1978) found concentrations (meq/100 g) of bases in the buried soil were commonly twice those of the ash material immediately above.

The effects of vegetation, parent material, and climate seem to be evident in these chemical data. Youngberg and Dyrness (1964) found snowbrush (*Ceanothus velutinus*), a known nitrogen fixer, enriched the surface 2 cm of pumice soils with N. Calcium was also higher in the surface. Geist and Strickler (1978) found vegetation-associated differences in bases among ash soils (L,M,SF in table 2), with those supporting spruce-fir being much lower. In contrast, organic matter and total N levels were much higher. Spruce-fir sites are associated with wetter and cooler conditions, which are conducive to leaching of bases and slowed decomposition of organic matter. Unfortunately, we cannot separate vegetation and climatic influences.

Basalt-derived soils supporting ponderosa pine in eastern Oregon had similar organic matter, total N, and extractable P levels, but were generally higher in extractable base levels. These soils were geographically associated with the ash soils supporting lodgepole pine and mixed conifer types, so macroclimatic influences should not differentially affect soil chemistry; but we still have confounded effects of vegetation and parent material.

Youngberg and Dyrness (1964) suggested soil chemical data should be compared on a mass per unit area (volume) basis to account for large disparities in bulk density. Comparisons of classical kinds of data such as in table 2 are valid among soils of similar bulk densities. For purposes of broader comparison we include values representative of the upper 60 cm (2 ft) of mineral soil for three macronutrients (table 3). These values further indicate pumice soils are on the low end of extractable bases and P range (as are ash soils under spruce-fir), but have moderate levels of nitrogen (depending on positional availability to root systems). Youngberg and Dyrness also noted the upper 60 cm of mineral soil supporting medium-site Douglas-fir in western Oregon could contain about 8,960 kg/ha N and proportionately higher levels of other nutrients compared to pumice soils. Differences in comparison with ash soils are not always this great for some nutrients, but there are some large differences in nutrient capital (Geist and Strickler 1978).

Our tabular data do not include contributions from organic horizons, which are often thin. However, litter layers are generally known to supply significant levels of nutrients. The contribution of nutrients from litter is probably more important now than historically, since fire

Table 2—Generalized chemical properties of ash-, pumice-, and basalt-derived soils (adapted from Geist and Strickler 1978; Youngberg and Dyrness 1964)

Vegetation[1]	Depth	Organic matter	Total N	Extractable P	Extractable Ca
	cm	*Percent*	*Percent*	*ppm*	*meq/100g*
		Ash-Derived Soils of Eastern Oregon			
L	0-15	3.60	0.09	51	4.2
	15-30	1.60	.05	31	4.3
M	0-15	3.43	.10	63	8.2
	15-30	1.93	.06	36	6.8
SF	0-15	7.89	.22	45	3.0
	15-30	4.70	.16	21	1.8
		Pumice-Derived Soils of Central Oregon			
P/B	0-2	6.41	0.14	12	2.4
	2-10	1.57	.03	6	1.2
P/S	0-2	9.54	.17	19	4.3
	2-10	1.23	.04	7	1.4
W/S	0-2	8.84	.23	20	6.1
	2-10	1.17	.03	9	2.0
		Basalt-Derived Soils of Eastern Oregon			
P	0-15	4.08	0.11	44	10.7
	15-30	2.16	.07	31	9.9

[1]L = lodgepole pine, M = mixed conifer, SF = spruce-fir, P = ponderosa pine, B = bitterbrush (*Purshia tridentata* [Pursh] DC.), in P/S and W/S the S = snowbrush (*Ceanothus velutinus* Dougl.), W = white fir or grand fir.

suppression and improved grazing practices have aided litter accumulations. Nielson-Gerhardt (1986) found substantial amounts of nitrogen mineralization in ash soils, as is noted later. Roots of ponderosa and lodgepole pine have been observed in litter layers of pumice soils. The presence of these roots may be further indication that nutrient supply from organic layers is important.

Potential nitrogen availability in volcanic ash soils before logging has been measured using laboratory assessments of nitrogen mineralization. Anaerobic test values averaged 9 ppm N for surface mineral layers in northern Idaho under mixed conifers (Niehoff 1985). Under mixed conifers in northwestern Montana, test values ranged from 10 to 16 ppm N in surface mineral layers and 48 to 93 ppm in organic layers (Nielson-Gerhardt 1986). In eastern Oregon and Washington, values for surface mineral layers of ash soils averaged 10 ppm under various conifer cover, about half the levels of basalt-derived soils (Geist unpublished data). Values for surface mineral layers of another group of 14 ash soils associated with several overstory types ranged from 8 to 44 ppm N and averaged 19 ppm (Geist 1977). These latter values were well above the average extractable ammonium N (7 ppm) and nitrate N (2 ppm).

Table 3—Generalized amounts per unit area, to 60 cm depth, of total nitrogen, extractable P and K in some ash-, pumice-, and basalt-derived soils (adapted from Geist and Strickler 1978; Youngberg and Dyrness 1964)

Nutrient	Ash-derived L[1]	Ash-derived MC	Ash-derived SF	Pumice P	Basalt P
			kg/ha		
Total N	1,973	2,157	5,263	1,992	2,292
Extr. P	110	132	94	31	105
Extr. K	1,135	1,521	755	614	1,458

[1]L = lodgepole pine, M = mixed conifer, SF = spruce-fir, P = ponderosa pine.

The foregoing discussions of physical and chemical properties indicate volcanic ash and mixed ash-pumice depositions represent a forest-growth-enhancing addition to the prior soil; but where pumice soils lack mixing in the C1 and C2 horizons (nearer the source), a less favorable growth medium exists after deposition.

HARVEST EFFECTS ON SOIL PROPERTIES AND TREE GROWTH

The potential for adverse effects on forest growth associated with mechanical impacts on soil properties has concerned both foresters and soil scientists for decades. Relatively few studies of mechanical impacts on ash and pumice soils of interior forests have been conducted; but the findings, so far, are similar to those obtained elsewhere on other soils.

Investigations of tractor harvesting on 11 units with ash soils in eastern Oregon and southeastern Washington showed that 10 to 36 percent of the area of logging units sustained excessive compaction. Excessive compaction was defined as a 20 percent increase in bulk density (Geist and others 1989). These data excluded area in

the transportation system but included skid trails and landings. Notably, there were some units where significant percentages of area were excessively compacted, but changes in average bulk density were relatively small.

Sullivan (1988) monitored harvest units to assess soil impacts from single and multiple forest management entries. He found eight of 13 units with ash soils had detrimental impacts significantly exceeding 20 percent of the areas ($P \leq 0.10$). Nearly all of the detrimental impacts were associated with excessive compaction.

Cochran and Brock (1985) investigated relations between soil compaction and tree growth in a deep, well-drained soil formed from Mount Jefferson ash 50 to 60 cm thick over residuum or colluvium. They found both total height and height growth of trees in 5-year-old ponderosa pine plantations decreased with increasing bulk densities. When contrasting high and low soil disturbance classes in silt loam ash soils of eastern Washington and Oregon, Geist and others (unpublished) found reduced height and radial growth of lodgepole and ponderosa pine associated with compaction and soil displacement.

In the mixed conifer type of northern Idaho, Nielson-Gerhardt (1986) took 0- to 10-cm samples of volcanic ash soils to assess the effects of logging disturbance on estimates of nitrogen availability using anaerobic incubation tests. Test values for samples from severely disturbed sites were 50 percent lower (5-10 ppm N) than samples from relatively undisturbed sites within the same clearcuts. Because the severely disturbed category was missing an organic layer, no mineralized N was available from that source. Severely disturbed areas constituted 25-50 percent of the four clearcuts studied. Organic matter differences were not detected, but a high degree of mixing and variability was associated with disturbances. On the average, ash thickness was cut in half (loss ranged from 10 to 15 cm) because of displacement and compression. The rooting medium for new tree seedlings was sharply reduced, because the underlying glacial till was ice compacted and restrictive to root growth. Severe disturbance categories had 19 to 58 percent higher bulk densities in the ash layer than unlogged forest controls.

Niehoff (1985) found no significant differences in organic matter content or nitrogen mineralization tests of volcanic ash soils except in the extremely burned category 1 year after clearcutting and burning mixed conifer stands in northern Idaho. The extreme category had lost all the organic layer and was pink to red 1 to 5 cm into the mineral surface layer. Less than 1 ppm N was mineralized in the extreme category; other categories, including a forested control, had 9 to 10 ppm of N mineralized from 2.5 to 7.5 cm samples of anaerobically incubated mineral soil.

In studies of ash-influenced soils of north-central Idaho, Clayton and others (1987) measured diameter, radial growth, and height of trees 15-25 years old in relation to soil displacement and compaction assessments. One or more growth variables were significantly reduced in response to changes in displacement or compaction disturbance classes. The trees were naturally regenerated lodgepole pine or planted ponderosa pine.

Compaction was found to persist for decades in volcanic-influenced soils studied by Froehlich and others (1985) and in silt loam ash studied by Geist and others (1989). Froehlich and others found persistence depended mainly on the initial degree of change, which in turn may be influenced in varying degree by soil texture, structure, moisture conditions, number of machine passes, loading, and operator skills. No difference was found in the rate of recovery between two different textured soils.

From the foregoing it is evident that, even though ash and pumice soils have relatively low bulk densities before and after disturbance, there is a strong association between growth loss and compaction. In addition, there is evidence to suggest other disturbance influences, such as displaced soil that removes onsite nutrients and water-holding capacity, may reduce growth. More attention is warranted, not only to compaction, but to these other influences.

RESPONSES TO FERTILIZATION

Pumice and ash soils in south-central Oregon have generally low fertility levels. Nitrogen (N), phosphorus (P), and sulfur (S) seem to be the limiting nutrients. Our view is that S and P appear to be more limiting to grasses than to trees, but our experience is limited to relatively few members of the ash and pumice soil populations.

Greenhouse studies of nutrient additions to ash soils (0- to 15-cm mineral depths) found strong N and S interaction responses in grass growth. There were indications that N additions without S could reduce growth (Geist 1971). Tests of elemental S versus calcium sulfate, in combination with either ammonium nitrate or urea sources of N, showed the sulfate source was clearly superior for stimulating growth of grass (Geist 1976a; Klock and others 1971).

Pumphrey (1971) annually fertilized a northeastern Oregon ash soil for 4 consecutive years with 67 kg N, 11 kg P, and 12 kg S/ha. Dry-matter production of 28 grass species was increased an average of 2,016 kg/ha. Growth began earlier and green period extended longer with fertilization. Geist (1976b) compared production of newly seeded orchardgrass (*Dactylis glomerata* L.) on ash soil using various fertilizers at a rate of 100 kg N/ha. First-year dry matter yield for the unfertilized control was 213 kg/ha; the N-only treatment from ammonium nitrate yielded 372 kg/ha, and the N plus S treatment from ammonium sulfate yielded 1,490 kg/ha. Subsequent research on ash soil fertility has shown varying degrees of N, S, and P deficiencies and thus variable nutrient interactions (Geist 1977, 1979).

When pine seedlings are grown in the greenhouse on material from the AC horizon of the Lapine soil (pumiceous, Xeric Vitricryands) in central Oregon, responses to N, P, and S and all the interactions were significant ($P \leq 0.05$, Youngberg and Dyrness 1965). Fertilizer ratios of N:P:S that appeared to produce the best seedling response in the greenhouse were 200:100:33. Boron (B) and other micronutrient deficiencies have been suspected, but greenhouse or field trials have not confirmed these suspicions. Thinned stands of ponderosa or lodgepole pine show significant responses ($P \leq 0.05$) in bolewood growth when fertilized with N, P, and S in the above ratios at rates of N ranging from 224 to 672 kg/ha (Cochran 1978, 1979, 1989). Lodgepole pine fertilized at the highest rate

produced 131 percent in gross periodic annual volume increment (PAI) over the controls during the first 4-year period after application. Ponderosa pine fertilized at the lowest level produced a 56 percent increase in gross PAI over the controls during the first 5 years. Much of this response appears to be related to N, and fertilization with N alone at a rate of 224 kg/ha produces a significant growth response. S appears to be the next most limiting nutrient (Will and Youngberg 1978).

A field study of N, P, and S combinations showed basal area growth of thinned ponderosa pine was superior using N plus P plus S compared to N plus S, N plus P, or N alone (Cochran 1978). Similar superiority was not found with bolewood, perhaps because it is more difficult to measure bolewood as precisely. Thinned white fir (*Abies concolor* [Gord. & Glend.] Lindl.) on pumice soils fertilized at rates of 224 kg/ha N plus 40 kg/ha S experienced more than a 32 percent increase in gross PAI over a 5-year study period (Cochran 1990). Thinned interior Douglas-fir responds well to fertilization with 224 kg/ha N on ash soils in eastern Washington and northern Idaho, but increasing the rate to 448 kg/ha N did not yield additional response. Similar results were obtained in thinned ponderosa pine on ash soils of eastern Washington and northeastern Oregon. Fertilized western larch in northeastern Washington produced 27 percent more gross PAI than controls during an 8-year period after fertilization (Cochran and Vander Ploeg 1988).

Grasses growing under pine stands on the pumice-mantled plateau also respond to fertilization. Growth of western needlegrass (*Stipa occidentalis* Thrub. ex Wats.), and bottlebrush squirreltail (*Sitanion hystrix* [Nutt.] J.G. Smith) under a thinned lodgepole overstory was significantly increased ($P \leq 0.05$) by fertilization with 672 kg N plus 336 kg P plus 111 kg S/ha (Cochran 1975a). Lower rates of nutrients in other studies on the pumice-mantled plateau also appeared to increase grass production, but the response was not quantified (Cochran 1978, 1979). Responses of the shrub understory to fertilization of pumice soils were inconsistent, apparently varying by individual plant. Leader growth of some bitterbrush (*Purshia tridentata* [Pursh] DC.) plants under either lodgepole or ponderosa pine appeared to respond to fertilization, but others did not. Measuring the response of bitterbrush plants in the field was further complicated by selective browsing of big game. Some bitterbrush plants were heavily browsed, while others nearby appeared untouched. Fertilized plots of Idaho fescue (*Festuca idahoensis* Elm.) in the understory of ponderosa and lodgepole pine were also heavily grazed by game, but control plots were hardly grazed at all. Geist and others (1974) found fertilization increased big game preference for grass growing in eastern Oregon ash soils.

CONCLUSIONS

Understanding the beneficial, detrimental, and in some cases, unique properties of volcanic ash and pumice soils is critical to maintaining the long-term productivity of interior forests in tephra deposition zones. Mismanagement of these soils can easily occur. If it does, full rehabilitation may not be possible, and productivity will be reduced.

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Speakers answered questions from the audience after their presentations. Following are the questions and answers on this topic:

Q.—What is the relative compaction hazard of ash cap soils compared to sands, silts, and glacial tills?

A.—There appears to be considerable similarity among textural groups in their response to compaction, if one expresses compaction in terms of percentage increase in bulk density. Thus, the absolute values of bulk densities before and after compaction are less important than the percentage change. This view follows that of Froehlich and McNabb (1984), who provided a more detailed explanation with appropriate qualifiers. It has been my feeling that volcanic ash soils compact quite easily (readily compact), but I haven't seen engineering studies that would bear out my viewpoint. Regardless, ash soils do not achieve particularly high bulk densities (are not highly compactable). I suggest you also consult Geist and others (1989).

Q.—Does compaction of soils make trees more susceptible to insect and disease?

A.—I believe this is the case. However, I have not conducted studies in this regard. There are other papers in this proceedings that may offer you specific answers. My view is that compaction (site stress) increases physiological stress in trees which in turn increases vulnerability to insect and disease.

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MONITORING FOREST SOIL PROPERTIES TO MAINTAIN PRODUCTIVITY

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T. A. Max

ABSTRACT

This paper addresses considerations necessary to construct a scientifically based sampling system for monitoring soils. A statistically sound method, which uses line transects, is described, and several applications for various sampling objectives are discussed. Results from sampling forest harvest units illustrate how data variability and choice of statistical precision levels affect sample size requirements. The transect system has been found to be an objective and easy-to-apply approach to monitoring forest soils in a variety of situations.

INTRODUCTION

Various Federal legislation directs natural resource management agencies to steward the public land in a manner that maintains or enhances productivity. The Multiple-Use Sustained Yield Act of 1960 called for management of renewable resources of the National Forests without impairment of land productivity, and more recently, the National Forest Management Act of 1976 emphasized protection and improvement of soil resources. The latter act also included instructions to monitor the effectiveness of management in meeting planned goals. Such monitoring includes soil resource monitoring.

Monitoring needs may vary relative to management objectives and the soil characteristics measured. To date, however, most forest soil monitoring has focused on alteration of physical properties. Ultimately, there is a need to translate soil status to site productivity—commonly measured as tree growth. To do this, research is needed to establish relations between specific changes in soil condition and plant growth. Managers, then, have two major jobs: detection of changes in soil characteristics through monitoring, and interpretation of the effects of these changes on site productivity based on growth or other measures provided by research. It is imperative that research scientists and land managers work together to improve detection of changes in soil characteristics and to translate these changes into effects on forest soil productivity. Improved assessment methodology could be developed as a part of monitoring, as an independent research effort, or a combination of both. In this paper, we will only discuss monitoring to detect changes in soil characteristics.

A sound monitoring system should contain several key elements to ensure the sampling system will accurately reflect the properties of the area(s) assessed. These key elements include:

1. Ability to objectively assess soil characteristics regardless of their spatial distribution.
2. Unbiased choices of sampling locations.
3. Well-distributed coverage of the monitored area.
4. Flexibility in the kind of information gathered and the tools that might be used.
5. Statistically valid sampling design to provide objective estimates of sampling error; statistical tests, if needed; and methods for controlling precision of estimates.

STANDARDS AND EXAMPLE SYSTEM

A transect system that includes these key elements and utilizes existing statistical theory for monitoring forest soils in the northwestern United States was developed and described by Hazard and Geist (1984). The rationale and steps used were later related to National Forest planning by Miller and Hazard (1988). The field and office procedures for conducting the monitoring were published as a "how-to" guide for soils specialists and other monitoring personnel of the Forest Service (Howes and others 1983). The transect system was subsequently adopted as the "standard" for monitoring soil physical conditions of National Forests in the Pacific Northwest Region, and formed the basis for nationwide soil monitoring methods adopted by the Forest Service in the National Soil Management Handbook (FSH 2509.18).

Various standards or guideposts are a necessary part of monitoring so the effectiveness of management prescriptions can be judged relative to provisions of forest plans. These standards relate to limits of overall operation or individual operational effects, which on the basis of biological or productivity knowledge reflect the tolerance of soil-vegetation systems to manipulations of various kinds.

The Pacific Northwest Region, and the Forest Service nationally, have adopted several standards for maintaining both long- and short-term soil productivity. The Pacific Northwest regional standards are based on the premise that productivity is a function of several state variables including available soil moisture, available soil nutrients, and soil aeration as discussed by Meurisse (1988). Further

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discussion about the soil capacity factors is included in these proceedings (Meurisse and others). The national standard for determining significant changes in productivity of the land is from a consensus of experts based on available research and current technology. National Forests are encouraged to establish standards of their own, if justifiable. The general standard for National Forests in the Pacific Northwest Region is: "a minimum of 80 percent of an activity area, including transportation system, should be left in a condition of acceptable (nondetrimental) productivity potential for trees and other managed vegetation following land management activities." Defined detrimental conditions are:

1. *Compacted*—an increase in soil bulk density of 20 percent or more in volcanic ash and pumice soils, or an increase in soil bulk density of 15 percent or more or macropore space reduction of 50 percent or more in other soils.
2. *Puddled*—depth of rutting is 6 inches or more.
3. *Displaced*—removal of 50 percent of the topsoil or humus-enriched A1 and/or AC horizons from an area 100 square feet or more which is at least 5 feet in width.
4. *Severely burned soils*—those where the top layer of mineral soil has been changed in color, usually to red, and the next half-inch blackened from organic matter charring.

More detailed descriptions of these standards are included in Meurisse (1988).

This monitoring system and standards have been widely used in National Forests of the Pacific Northwest, primarily in monitoring soil compaction and displacement associated with timber harvesting. Sullivan (1988) monitored other effects as well. Briefly, the system uses a systematic grid of points that is randomly positioned over the area to be sampled. Randomly oriented transects radiate from each grid point (fig. 1). Surface and subsurface monitoring information is gathered by categorizing segments of each transect line according to soil condition and by collecting soil samples at intervals along the line. A 100-foot transect length is used. As a result, summed segment lengths, expressed in feet, are percentages of transect lines in a given soil condition. The average, by condition, over all transects represents the percentage of area in that condition for the monitored unit. Other line lengths can be used, so long as measurements are converted to percentages, but we have found that 100 feet is an adequate length. Soil condition percentages by line are used to calculate statistical attributes of the sampled area, such as means, standard deviations, and standard errors. Interval samples along the transects are used for assessments like soil compaction, for which intact cores or penetration readings may be taken. The system can also accommodate fertility samples. In the case of compaction, core samples are extracted from a specific depth or soil horizon at 10-foot intervals beginning at the 5-foot mark. Bulk density values are determined and then compared to the average bulk density of additional random transects (at least three) located on an untreated area (one which provides an appropriate predisturbance background value), probably adjacent to the sampled area. The average value of bulk density from transects in undisturbed areas plus 15 or 20 percent (that is, undisturbed average multiplied by 1.15 or 1.20) represents the lower limit of unacceptable or detrimental degree of compaction. Each core sample represents 10 percent of each transect length. Thus, if three core samples of a line transect exceed the bulk density limit, then 30 percent of the line, or area the transect represents, has been detrimentally compacted.

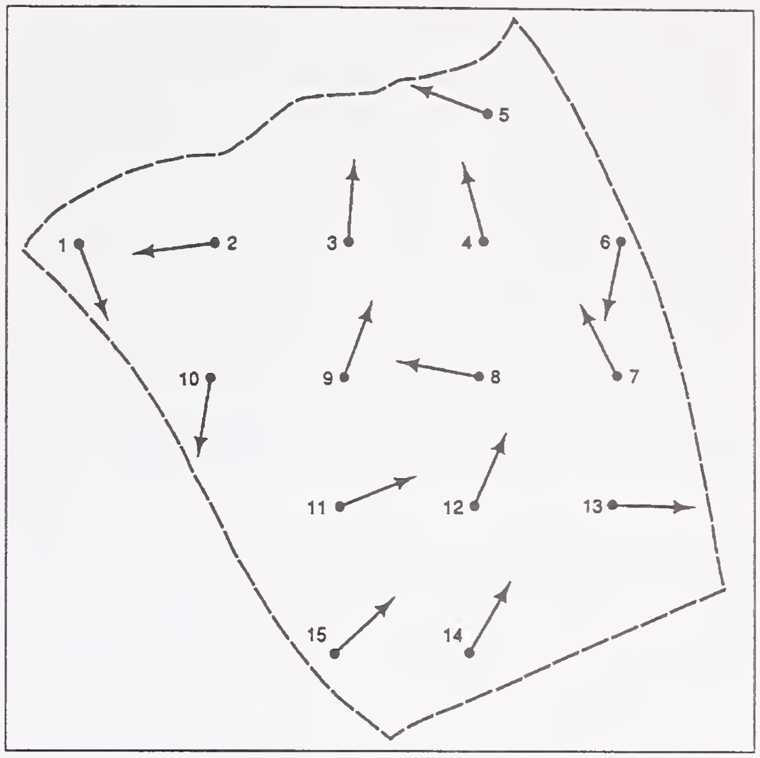

Figure 1—An example of the randomly positioned, systematic grid with randomly oriented transects.

The soil condition categories monitored by practitioners in the Pacific Northwest vary somewhat, but they include slash piles, landings, skid trails, spur roads, compacted, displaced, puddled, eroded, miscellaneous, deposited, and undisturbed. Categories can readily be changed, so they apply to a wide variety of situations. The latter two categories are not considered to be detrimental.

Two computer programs were developed to save computation time. One calculated bulk densities from core measurements (Starr and Geist 1988), and the second summarized the data and computed statistical estimates (Hazard and others 1985). The calculated variance for overall detrimentally affected area provides users a statistical test for whether general limits of total damage were exceeded. A statistical test may not be needed or desired in some cases, but this option is available. The same program can compute estimates of numbers of samples (number of transects) needed to satisfy specified precision requirements. We present example data from using this option later in the paper.

APPLICATIONS

There are a number of ways this sampling scheme can be applied in meeting monitoring needs, and this flexibility increases its usability for land managers. We present some examples here.

Single-Activity Area Evaluations

To date, this system has primarily been used to estimate the final status of soil characteristics for an individual area, like one unit of a timber sale (fig. 1). Repeated application of the sampling design on the same unit can be used to assess the effects of a series of activities either individually or collectively (that is, cumulative effects). Repeated measures optimize sensitivity for detecting change by reusing the same gridpoints and transects at each sampling time. When performing repeated measurements, slight offsets are required where destructive samples are obtained; the procedural decisions for selecting the offset positions are part of the rules established before fieldwork begins.

Comparisons Among Areas

Sometimes the question arises: Are the average values for soil conditions significantly different among units? Statistics can help handle the variability that often confuses our ability to otherwise make objective comparisons. Monitoring is carried out on each of the units using the same sampling approach as for a single unit. The percentages by line are the observations used to statistically test differences among units. An unpaired t-test would be an appropriate statistical test to contrast two units. Comparisons among more than two units would require a different statistical approach (for example, a multiple comparison procedure) to properly account for multiple nonindependent statistical tests.

A word of caution is noteworthy here. Comparisons among single-unit means would not suffice as a comparison of different management activities or treatments that happened to be associated with each mean. This is because no measure of variability exists for the activities. We address activity comparisons next.

Sampling to Compare Differences Among Activities

Where a comparison of the effects of two (or more) kinds of activities on soil properties is desired (for example, random skidding vs. designated skid trails), sets of management units are chosen for each practice. Activities are assumed to be applied to the whole unit. Each set of units represents the range of conditions over which one of the activities occur (the sample of units represents a well-defined population of interest). The procedures for transect-sampling each area are the same as described for an individual unit. The unit average of transect percentages represents a single observation, a replicate, in the array of unit averages for a soil condition associated with the activity. The two arrays of averages are used for statistically comparing activities. A one-way analysis of variance would be an appropriate statistical test. The number of units used to assess the effects of each activity should be similar but not necessarily equal. Sensitivity of the test to detect differences is increased as the number of units (n) increases, but also depends on the magnitudes of the true differences among activities. Auxillary information to identify differences among units, independent of the activity, may be used in covariance analysis or blocking to increase the sensitivity of the statistical comparison.

Stratified Sampling

Stratification can increase the efficiency of sampling. In general, improved efficiency occurs when a population can be divided into strata so that the mean values (of the variable of interest) differ among the strata, yet strata are relatively homogeneous. The transect system can be used here too, but the user may need to apply procedures for shifting transect positions because they cannot cross strata boundaries.

Sampling soil compaction is an example where stratification might work well. Much of the compaction occurring on a logged area may be concentrated in rather easily discernible skid trails. In such a case, two strata would be defined as skid trails and nonskid trails. The expectation is that the percentage of area compacted on skid trails is much higher than for the rest of the unit. Variation within each strata would be less than variation across the unit as a whole. Hence, we expect stratified sampling to be more efficient in this case, if it is not too expensive to determine strata boundaries.

Part of the increased efficiency of stratified sampling results from more information about the population being included in the estimators. In particular, strata sizes (or at least their relative sizes) must be known, which requires area determinations. Areas must be measured easily and cheaply for stratified sampling to be efficient.

A remedy for transects crossing boundaries would be to reorient them by choosing a new random direction for each. Other remedies might include a different grid size, a shorter transect length, or a different transect configuration for the skid trail stratum. Such solutions may require additional research to verify statistical adequacy.

Double Sampling

Core sampling is an accurate method of evaluating soil compaction, but it is too time consuming to meet the rapid survey needs that commonly arise. In such cases we would encourage exploratory tests of rapid surveys using double sampling. For double sampling, a large and a small sample are taken. The small sample is taken using a standard, more time-consuming method. The large sample is taken using a faster, less precise method. The small sample is a subset of the large sample, hence the term double sampling. For example, we will use core sampling and probing with a sharpshooter shovel as the two methods to assess compacted percentages of transects (still the basic sampling unit). The sampler must first obtain a "feel" for excessively compacted conditions in a controlled area where conditions are known. The sharpshooter measurements would be categorical, that is, classified above or below the specified detrimental compaction limit, rather than continuous values. Once an individual develops the needed feel for a given soil, rechecking calibration will require less core sampling. In this application, the large sample is taken by probing sample points along all transects on the area to determine the compacted percentage of each transect. The small sample, of transects selected systematically or randomly from all transects on the area, is taken by core sampling at the same sample points, offset to avoid disturbance from the shovel. Compaction percentages of the subsample transects are then computed from

the core sampling data. The correlation between transect proportions, as determined by the two methods for individual lines in the subsample, as well as the relative costs of the two measurement techniques, determine the efficiency of double sampling and the optimal size of the small sample.

Details about the theory of double sampling are covered by Cochran (1977). Some helpful discussions on the use of double sampling are given by Geist and Hazard (1975), even though the application is different. Another potential application of double sampling was reported by Clayton and others (1987), who used a penetrometer and sharpshooter shovel to classify compaction.

RESULTS FROM SINGLE-AREA SAMPLING

The transect sampling system was developed in cooperation with the Pacific Northwest Region of the Forest Service, and the Umatilla, Malheur, and Wallowa-Whitman National Forests. One objective was to assess the amount of detrimental conditions (excluding transportation system in this case) for an array of 11 harvest units (Geist and others 1989). These efforts tested the system, gave useful information on harvest effects, and initiated a data base for the variability of soil characteristics. This information provides a basis for estimating numbers of samples at prescribed levels of precision at specified statistical probabilities. For example, the number of transects (grid points) needed to determine the average detrimentally impacted area within 20 percent error at two levels of probability are shown in table 1. More transects are necessary when variability is higher, when a higher probability is desired (90 percent vs. 80 percent) or when a smaller percentage of error is desired. An 80 percent probability translates to a 1 in 5 chance of error; a 90 percent probability means a 1 in 10 chance.

Table 1—Number of transects required to estimate average detrimental soil conditions within 20 percent of the true mean percentage, at two probability levels for 11 harvest units (adapted from Geist and others 1989)

Harvest unit	Summary statistics[1] $\bar{X}$	s	Probability level 0.80	0.90
	Percent of area		*No. of transects*	
1	25	20	27	45
2	21	25	58	95
3	44	25	13	21
4	37	17	8	13
5	27	14	12	20
6	23	18	27	43
7	21	15	21	35
8	19	15	24	40
9	34	22	17	28
10	41	15	5	9
11	24	16	20	33

[1]Mean and standard deviation, respectively, of percent area in detrimental condition.

Sullivan (1988) used this monitoring system to assess soil physical conditions following tractor logging and machine piling in 24 timber sale units in the Malheur National Forest in Oregon. He determined the "Average Percent Detrimental Impact" (APDI) and found that 15 of the units exceeded the general regional standard of 20 percent area. He summarized results in various ways: by general soil type, by timber type, by silvicultural prescription, and by previous activity. The latter involved repeated measures to determine cumulative effect. Some 95 percent of the APDI detected was from compaction. The results of his sampling have been used to take preventive or remedial measures including reducing the degree of slash disposal, using designated skid trails approved prior to tree felling, and using a winged subsoiler to treat portions of units that exceeded the general standard.

In a spinoff application of the sampling system, Cochran and Brock (1985) used the point grid and transect approach to study soil compaction and growth of young trees planted in two clearcuts. Growth measurements of trees within 1.2 m of the transects were compared to soil bulk density near the tree. Tree growth and bulk density data were summarized on a transect basis, and regression analysis was used to test relations between variables.

CONCLUSIONS

The sampling system discussed here should not be considered the last word in strategies for soil monitoring; however, it is flexible, objective, and easy to apply. It is hoped these pioneering efforts will evolve into progressively better sampling methods. If other applications of this system are explored, statistical consultation should be sought to help meet needs and retain statistical validity.

Those specialists who become involved in monitoring natural resources often find knowledge gaps that limit their ability to measure and interpret changes in productivity or other environmental attributes. Some knowledge gaps will not be filled soon, and those who seek to fill them will find a challenging arena for investigation because of the complex interactions and frequently confounded effects of management on ecosystems. Studies relating adverse soil conditions to plant growth have commonly been after-the-fact observational studies rather than designed experiments. Some future research should be designed to quantitatively define cause-effect relationships. The results would help guide monitoring to maintain forest soil productivity.

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ORGANIC MATTER FUNCTION IN THE WESTERN-MONTANE FOREST SOIL SYSTEM

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ABSTRACT

Soil organic horizons are critical components of forest soil productivity. Understanding their unique roles in moisture retention and nutrient cycling before and after timber harvesting is key to managing postharvest productivity for future stands. Diverse habitat types, variable volcanic ash deposition, and temperature/moisture extremes make organic horizons especially crucial to productivity and management of western-montane forests.

INTRODUCTION

Productivity of western-montane forest soils is tightly bound to the organic matter component. Additions, alterations, and reductions of forest litter, humus, and wood residues influence both biotic and abiotic properties of any given site (Harvey and others 1987). Organic matter is especially important for soil water retention, cation exchange, nutrient cycling, and erosion control. Recent trends toward whole tree harvesting, increased woody debris removal (including slash burning and yarding unmerchantable material), and shorter rotations (McColl and Powers 1984) have increased awareness of effects of forest operations on soil processes and overall site productivity (Jurgensen and others 1990).

FOREST SOILS

The western-montane area encompasses the area from the eastern slope of the Cascade mountain range in Washington and Oregon south to the Sierra crest in California. It extends east to the Continental Divide in Montana and Wyoming.

Most forest soils in the montane-west are Inceptisols (some Andosols, under the new taxonomy), developed from volcanic ash deposits (see Meurisse and others and Hironaka and others, these proceedings). The major ashfall affecting the area is from Mount Mazama (now known as Crater Lake, OR) ash, which was deposited during an eruption 6,700 years B.P. The western-montane region was blanketed with a highly variable, sometimes extensive ash-fall up to 60 cm deep. Lesser eruptions from Mount St. Helens and Glacier Peak left thin deposits of ash.

Ash cap soils support large tracts of forested land in the Inland Northwest and have relatively high productivity and water-holding capacity compared to non-andic soils (Geist and others 1989). Below the ash cap in northern Washington, Idaho, and Montana is glacial outwash material characterized by a high percentage of rock fragments and low moisture and nutrient-holding capacity. Andic soils, once the organic mantle has been removed, are very susceptible to damage, particularly erosion, during harvest and site preparation.

The litter layer, also designated as Oi, consists of freshly fallen needles, twigs, and other debris that have undergone only slight decomposition. The fermentation layer (duff or Oe), is the organic material beneath the litter layer. Decomposition in this layer is very active, and the duff is usually permeated with fungal mycelia and root mats. Although this horizon is undergoing decomposition, plant parts are still distinguishable. Humus (Oa), is unrecognizable, dark brown or black, amorphous organic material that has undergone complete decomposition.

Typical horizon development includes some form of an organic horizon (O) underlain by an A and a Bs horizon (Fosberg and others 1979). Relatively undisturbed surface organic horizons typically consist of approximately 2-10 cm of litter, 0-5 cm of duff or humus (fig. 1) (collectively termed the forest floor), and varying amounts of decayed soil wood (the brown, crumbly mass left from decaying wood) (Harvey and others 1987). Surface organic horizon depth is highly variable, depending on climate, moisture, and topography. Northern Idaho forests, which are cool and moist, generally have substantial organic deposits, except in the driest habitat types. Central Idaho and northwestern Montana have similar organic horizon depths. However, soils influenced by lodgepole pine (*Pinus contorta* Dougl. ex Loud.) on warm, dry sites in central Idaho have almost nonexistent organic horizons and reflect a lower productivity potential (Steele and others 1981).

Woody residue is a valuable component of montane-west ecosystems and has an important role in carbon cycling, nutrient storage, stream dynamics, erosion control, and animal activity (Harmon and others 1986; Jurgensen and others 1990; Maser and Trappe 1984). When the woody residue becomes incorporated into the forest floor, it is then termed soil wood. Organic horizons in combination with woody residues and soil wood comprise most forest soil

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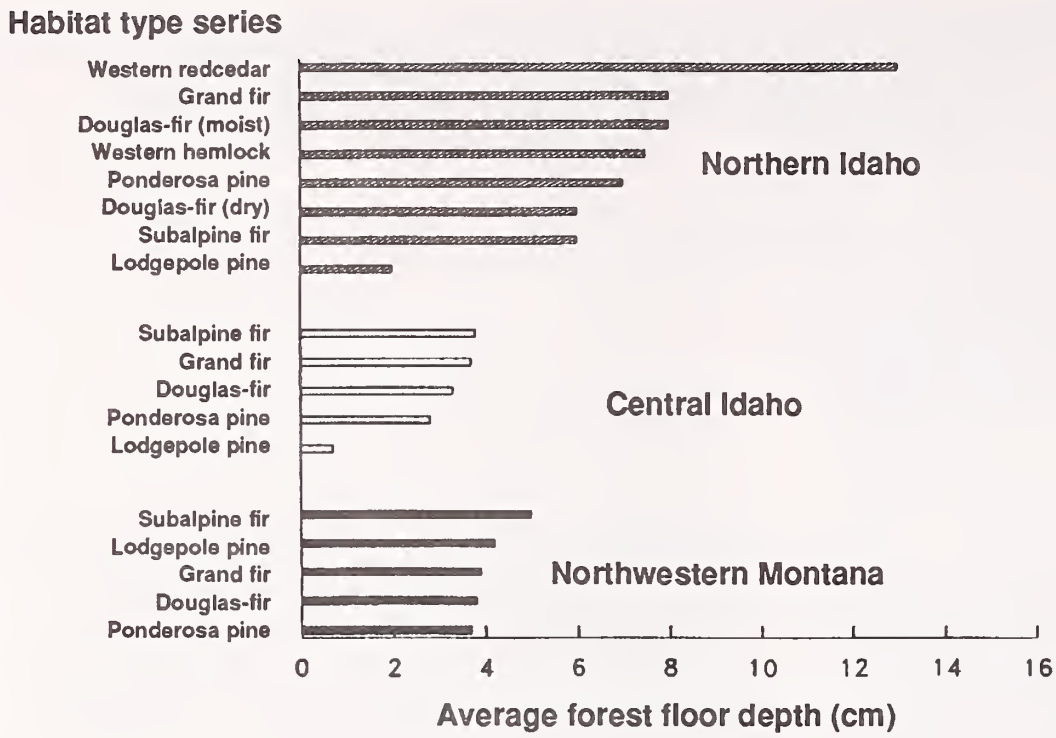

Figure 1—Average depth of the forest floor in selected habitat types in northern (Cooper and others 1987) and central Idaho (Steele and others 1981), and northwestern Montana (Pfister and others 1977) forests.

organic matter (table 1), and in many cases the woody residue component may equal or surpass that of other soil components.

Total soil organic matter contents generally mirror site productivity. The most productive stands in our region have the deepest organic matter deposits and are usually in the cedar/hemlock (*Thuja plicata* Donn ex D. Don and *Tsuga heterophylla* [Raf.] Sarg.) types. The least productive stands, with the shallowest organic matter deposits, are ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) stands. The exception to this rule is subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.) types; in these stands low temperatures limit organic matter turnover rates, leading to deep organic matter deposits, but limited tree growth.

Organic Matter and Nutrient Budgets

A primary portion of the nutrient capital, particularly nitrogen (N), in the forest ecosystem is contained in the Oi, Oe, Oa, and woody residue. Soils with the greatest N content usually have the largest organic horizon accumulations (tables 1 and 2). Generally, as N in the organic horizons increases, stand productivity increases. The exception to this is the warm, moist cedar/hemlock stands in Idaho where there is a rapid turnover of forest floor (Jurgensen and others 1990). In southeastern Wyoming, lodgepole pine (*Pinus contorta* ssp. *latifolia* [Engelm. ex Wats.] Critchfield) stands average 31 Mg/ha forest floor (Oe and Oi) volume and have an average N content of 33 kg/ha (Fahey and others 1985). On these stands, woody residue contributed 13 kg/ha N. Since lodgepole pine stands are usually N limited (Fahey and others 1985), inputs from decaying wood and forest floor can be very important for productivity.

Besides N, nutrients like calcium (Ca), magnesium (Mg), potassium (K), and phosphorus (P) are also found in abundance within organic horizons (table 3). The availability of all these nutrients is strongly influenced by the rate of

Table 1—Volume of organic horizons and mineral soil in old-growth stands in the montane-west (from Jurgensen and others 1990)

Location	Residue	Forest floor	Soil wood	Mineral soil[1]	Yield capability
	Mg/ha				*m³/ha/yr*
Montana					
Cedar/hemlock	84	50	51	145	7.7
Subalpine fir	146	36	36	153	7.7
Douglas-fir	45	26	26	133	4.9
Ponderosa pine	<20	7	2	160	2.9
Idaho					
Cedar/hemlock	154	23	48	201	9.5

[1]Sampled to a depth of 30 cm.

Table 2—Nitrogen content of organic horizons and mineral soil in several old-growth stands in the montane-west

Location	Residue	Forest floor	Soil wood	Mineral soil[1]	Proportion in mineral soil
	kg/ha				*Percent*
Montana[2]					
Cedar/hemlock	125	787	341	1,729	58
Subalpine fir	219	570	344	1,686	60
Douglas-fir	68	438	419	2,183	70
Ponderosa pine	<30	128	33	3,433	94
Idaho					
Cedar/hemlock	231	179	297	3,045	81
Douglas-fir[3]	ns	248	ns	3,160	
Wyoming					
Lodgepole pine[4]	ns	400	86	5,270	

ns = not sampled.
[1]Sampled to a depth of 30 cm.
[2]From Jurgensen and others 1990.
[3]From Clayton and Kennedy 1985.
[4]From Fahey and others 1985.

Table 3—Soil nutrient budgets of organic horizons and mineral soil from selected undisturbed stands in the montane-west

Horizon	Ca	Mg	K	P
	kg/ha/yr			
Ponderosa pine/Douglas-fir mixed forest-Silver Creek, ID[1]				
Litter (Oi)	347	340	340	190
Mineral (0-10 cm)	319	111	184	175
Lodgepole pine-Lolo Pass, MT[2]				
Forest floor (Oi,Oe)	349	48	120	100
Mineral (0-10 cm)	278	40	177	100

[1]From Clayton and Kennedy 1985.
[2]From Entry and others 1987.

Table 4—Moisture content of woody residue and mineral soil in the montane-west

Location	Woody residue	Mineral soil
	Percent dry weight	
Southwestern Oregon[1]		
Ponderosa pine	157	6
Western Montana[2]		
Douglas-fir	98	17
Subalpine fir	163	34
Hemlock	161	27

[1]From Amaranthus and others 1989.
[2]From Harvey and others 1979.

organic matter decomposition. Again, nutrient concentrations vary depending on overstory species and stand locations, but O horizons provide a large proportion of nutrients critical for seedling establishment and growth.

Organic Matter and Moisture-Holding Capacity

As we have seen, forest floor material and decayed logs are a reservoir for nutrients. They also act as a storehouse for moisture. Fallen, decaying logs can contain especially large amounts of moisture (table 4). Amaranthus and others (1989) noted, in southwestern Oregon, that during the winter months decayed wood acts like a sponge to absorb water and retains much of that water throughout the following growing season. This water supply can be particularly important for seedling establishment, especially where available soil water would otherwise be insufficient for surviving summer drought or for maximizing growth in highly competitive situations.

Comparisons of moisture contents on a dry weight basis do not provide a ready measure of how much is available for plant uptake. However, field capacity and permanent wilting point moisture data for a Douglas-fir stand in northern Idaho show soil wood has 5.5 times more available moisture than mineral soil per gram of substrate. On a weight/weight basis soil wood has an average available moisture of 84.5 percent, litter 18.7 percent, and mineral soil 15.4 percent (Page-Dumroese 1990). Although soil moisture levels fluctuate seasonally, decayed wood maintains higher water contents throughout the growing season (table 5) than the forest floor or underlying mineral soil. This makes decayed wood of particular importance to drier ecosystems where moisture is limited throughout the year. The forest floor, by acting as a mulch, may also be helpful for maintaining moisture levels in the mineral soil throughout the growing season.

Organic Matter and Cation Exchange Capacity

Organic matter, because of its many negatively charged sites, is a major source of a soil's cation exchange capacity (CEC) (Tate 1987). In a northern hardwood forest, Brooks (1987) found that in uncut stands the forest floor had six

Table 5—Seasonal moisture content fluctuations in soil substrates from an *Abies lasiocarpa/Clintonia uniflora* habitat type in western Montana (from Harvey and others 1978)

Month	Humus	Woody residue	Mineral soil
	Percent dry weight		
May	130	204	37
July	74	118	27
September	141	244	40

times greater CEC than surface mineral soil. After harvesting, an eightfold difference occurred in CEC between the forest floor and the mineral soil.

In northern Idaho, site preparation treatments that mound the soil organic matter and mineral top soil together (Page-Dumroese and others 1986, 1989) had significantly greater CEC's than a scalp treatment that removed the forest floor (table 6). The undisturbed treatment, with the forest floor left relatively intact, had a similar CEC to the mounded treatment. While knowledge about a soil's CEC is important, very little work has been done to link the effects of timber harvesting/site preparation to changes in CEC and resulting site productivity.

Organic Matter and Disturbance

Stand disturbances, either natural or artificial, have a dramatic impact on the depth of organic horizons (table 7). Recent wildfires and intense, long-duration prescribed burns seem particularly devastating to organic matter depth (Harvey and others 1986). Destruction of soil organic horizons by repeated wildfires over the past 75 years may be a contributing factor to the development of aggressive shrubfields in northern Idaho (Harvey and others 1987).

Harvesting and different site preparation methods and their effect on stand nutrient balances can be seen in table 8. Clearcut and burn operations maintain more total N, P, and cations in the organic horizons than does a mechanical residue (bulldozer piling) removal system. Acceleration of nutrient loss and increased erosion occur after removing the protective organic mantle (Megahan and Kidd 1972). Soil organic matter promotes the formation of water-stable aggregates as long as substantial levels are maintained. Once

Table 6—Cation exchange capacity (cmol/kg) and soil organic matter content (percent) as affected by site preparation technique in two northern Idaho stands

Site treatment	Low elevation[1]		High elevation[2]	
	O.M.	CEC	O.M.	CEC
	Percent	*cmol/kg*	*Percent*	*cmol/kg*
Mounded	15	15	28	18
Scalped	9	8	15	11
Undisturbed	14	11	29	20

[1]*Abies grandis/Symphoricarpos albus* h.t., elevation 715 m.
[2]*Tsuga heterophylla/Clintonia uniflora* h.t., elevation 1,456 m.

Table 7—Disturbance effects on soil forest floor (Oi, Oe, and Oa) depth in northwestern Montana (from Harvey and others 1986)

Site and disturbance	Stand age	Forest floor depth
	Years	*cm*
Subalpine fir		
Undisturbed	>250	3.5
Wildfire	80	1.8
Clearcut and burn	15-20	1.5
Wildfire	15	.5
Douglas-fir		
Undisturbed	>250	2.3
Partial cut/underburn	80-100	.6
Selective cut	60-120	1.2
Wildfire	50	1.9

Table 8—Forest floor and woody residue nutrient content left on an *Abies lasiocarpa/Xerophyllum tenax* site after harvesting and site preparation in northwestern Montana (from Entry and others 1987)

Harvest method	Ca	Mg	K	P	N
	- - - - - -	- - - - - -	*kg/ha*	- - - - - -	- - - - - -
CC[1]/residue left	331	46.1	79.7	145	634
CC/residue removed	188	26.8	40.8	60	392
CC/residue burned	215	27.3	75.4	10	476

[1]Clearcut.

the forest floor is destroyed, these aggregates break down and erosion increases. Clayton and Kennedy (1985) indicated it may take more than 50 years to restore a heavily disturbed ecosystem to its former nutrient status and perhaps centuries to restore soil lost through erosion.

Organic Matter and Regeneration

Postharvest natural and artificial regeneration success depends, in many cases, on soil organic matter content. Increases after harvesting and site preparation in organic matter percentage in the surface mineral soil are most likely the result of forest floor and a considerable amount of logging slash being mixed into the surface mineral soil.

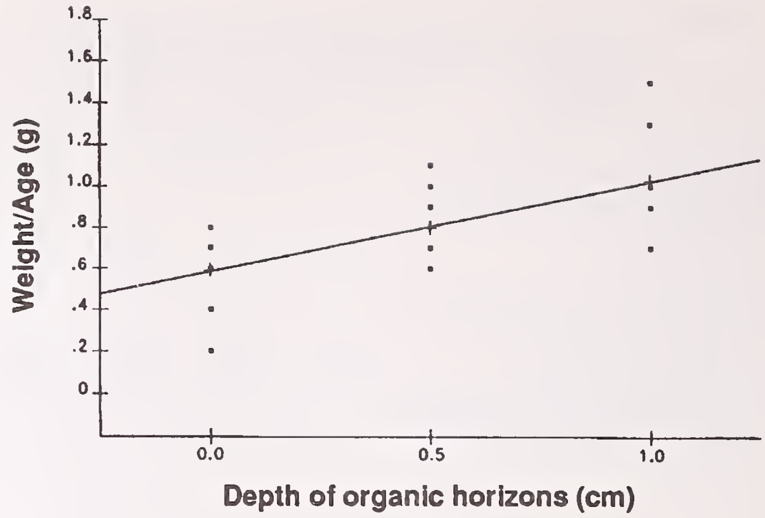

Figure 2—Ponderosa pine seedling weight response to increasing forest floor depth in the montane-west.

This increase is usually short-lived (Cromack and others 1979) and decreases with new stand development (Kraemer and Hermann 1979). Therefore, planted seedlings, with reduced access to soil organics, may have, or will likely soon experience, growth declines (Graham and others 1989).

Organic horizon depth can directly influence seedling biomass production (fig. 2). Seedling weight of naturally regenerated ponderosa pine in a Douglas-fir (*Pseudotsuga menziesii* Beissn. [Franco]) habitat type is positively correlated with depth of the organic horizon. The correlation shown in figure 2 is particularly striking because organic matter depth did not exceed 1 cm. Although this study had a relatively small sample size, organic matter depth explained 51 percent of the variation in weight of these seedlings (Harvey and others 1988).

In the past, mineral seedbeds for natural regeneration have been the "norm" (Haig and others 1941). However, soil organic components can also act as valuable seedbeds for natural regeneration (table 9). Organic substrates in the Canadian Rockies occupy a large portion of the stand and are used extensively as a seedbed.

Harvey and others (1987) noted that, in terms of a competitive advantage, conifers seem to be the only species using woody debris as a substratum for regeneration. There is also species differentiation in the use of organic horizons for regeneration (Day and Duffy 1963). Lodgepole pine favors a mineral seedbed, but Engelmann spruce (*Picea engelmannii* Parry ex Engelm.) and Douglas-fir prefer organic seedbeds. Organic horizons and the upper 30 cm of mineral soil then become the primary rooting substrate as seedlings mature (Harvey and others 1986; Kimmins and Hawkes 1978).

Growth of planted seedlings after intensive site preparation on two soil types in northern Idaho was influenced by soil organic matter content (table 10). Western white pine (*Pinus monticola* Dougl. ex D. Don) and Douglas-fir growth was greater after 3 years in treatments with high organic matter content compared to scalped treatments. This may be due to several interacting factors including:

Table 9—Seedbed composition and natural seedling distribution in logged-over stands in the Canadian Rockies (from Day and Duffy 1963)

Seedbed	Area	Seedling distribution
	Percent	
Muck	[1]—	9
Litter	11.8	9
Moss	6.3	24
Decayed wood	16.5	24
Humus	44.9	12
Mineral	20.4	22

[1]Data for area not available.

Table 10—Soil organic matter content and 3-year-old western white pine and Douglas-fir seedling biomass after three site preparation techniques in northern Idaho (from Page-Dumroese and others 1986; Graham and others 1989)

Treatment	Organic matter	WWP biomass	DF biomass
	Percent	Grams	
Mounded	27	19	16
Scalped	14	8	7
Undisturbed	23	8	9

(1) organic matter on the surface of the mounded treatments may have acted as a mulch to enhance water retention, (2) organic matter incorporated into the mounded treatments significantly lowered soil bulk density, and (3) organic matter left on the surface or incorporated into the mounded treatments improved the nutrient status of the soil (Page-Dumroese and others 1986, 1990). Scalping, which is commonly used in the montane-west, can in some instances, benefit seedling establishment and survival by reducing competition (Sloan and Ryker 1986). However, removal of the surface organic and mineral horizons can also severely limit growth and impair long-term survival.

MANAGEMENT IMPLICATIONS

Soil organic matter affects the cation exchange capacity, water-holding capacity, bulk density, nutrient budgets, and erosion potential. Removal of organic horizons during harvesting and site preparation may seriously reduce overall site productivity, stability, and regeneration potential.

Postharvest treatments should be planned to limit damage to fragile organic horizons. There may be occasional instances of extreme competition or heavy fuel loading that warrant intensive site treatments and forest floor removal to achieve adequate regeneration. Although maintenance of the organic mantle may limit some initial site preparation options, in the long run productivity will be maintained or improved. Economic investments made to conserve organic matter or reduce bulk density in many stands in the montane-west can provide substantial returns in the form of improved long-term soil productivity.

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NITROGEN FIXATION IN FOREST SOILS OF THE INLAND NORTHWEST

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ABSTRACT

Significant amounts of soil nitrogen (N) are lost from the soil during timber harvesting and related activities. Symbiotic N-fixing plants have the potential to replace much of these N losses on many sites in the Inland Northwest, especially during early stand development. However, many of these plants are site specific and can cause competition problems during stand establishment. Much more information is needed on the successional roles of N-fixing plants in Inland Northwest forests. Nonsymbiotic N fixation in forest soils of this region is low, but is an important source of N on sites where N-fixing plants are lacking or of low frequency. Appreciable amounts of N can be added to the soil by nonsymbiotic N fixation over long stand rotation ages typical for this region. Silvicultural systems need to be developed that minimize soil N losses and maintain the biological N fixation potential of the site.

INTRODUCTION

Timber harvesting and subsequent site preparation methods used in western forests can cause significant reductions in the levels of nitrogen (N) in forest soils (Cromack and others 1979; Page-Dumroese and others, these proceedings). The use of prescribed burns could greatly increase such soil N losses (Little and Klock 1985; Little and Ohmann 1988; Macadam 1987; Wells and others 1979). Nitrogen is required for tree growth in greater amounts than any other mineral nutrient, and is usually the nutrient most limiting in western forest soils (Edmonds and others 1989). Nitrogen is unique among the soil nutrients because it is present almost entirely in organic forms. No inorganic soil reserve is normally present to alleviate losses of N due to natural or human-caused factors (Wollum and Davey 1975).

Replacement of soil N lost due to forest management practices or wildfire in the Inland Northwest can come from four sources: (1) N present in precipitation and dry deposition, (2) biological N fixation by microorganisms living in plant roots (symbiotic N fixation), (3) N fixation by free-living soil microorganisms (nonsymbiotic N fixation), and (4) N fertilizers. The contribution of each will vary depending on forest age and vegetation type, site location, and management practices. Rainfall can add in excess of 20 kg/ha/yr to sites influenced by industrialized areas, but amounts normally average between 0.5 and 2 kg N/ha/yr in the Inland Northwest (Clayton and Kennedy 1985; Fahey and others 1988; Tiedemann and others 1978). Although some N fertilization trials have been conducted in this region (for example, Graham and Tonn 1985; Shafii and others 1989), widespread fertilizer applications are not presently considered economically feasible.

Symbiotic N fixation has the potential to add significant amounts of N to forest soils of the Inland Northwest. Nitrogen-fixing plants are amenable to manipulation by forest managers and could be considered when developing silvicultural prescriptions, especially those emphasizing forest biodiversity. However, as will be shown, many N-fixing plants are quite site specific, and can cause considerable problems for conifer regeneration. Nonsymbiotic N fixation rates in forest soils are quite low, but this N source is likely critical on sites where N-fixing plants are lacking.

SYMBIOTIC NITROGEN FIXATION

Nitrogen-fixing plants found in forests of the Inland Northwest are grouped into two categories: (1) plants in the family Leguminosae—nine genera, and (2) nonleguminous plants—five genera from four different families (table 1). The distribution patterns of these N-fixing plants generally reflect soil moisture/temperature conditions and stand successional stage. Early seral forests usually have greater shrub and herb development than late seral-climax forests, and would give a different picture of N-fixing plant distribution and importance.

Late Seral-Climax Stands

Legumes are more widely distributed than nonleguminous N-fixing plants in late seral-climax stands throughout the Inland Northwest. Studies in Idaho, Montana, and Wyoming have shown the genus *Lupinus* to be the most common N-fixing plant across all habitat types, while species of *Oxytropis* and *Lotus* were found only in scattered locations (tables 2 and 3). *Shepherdia* was the most frequently occurring nonleguminous N-fixing plant, although *Alnus* and *Purshia* were common in certain habitat types.

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Table 1—Nitrogen-fixing plants occurring in Montana, Idaho, and western Wyoming forest habitat types[1]

Nonleguminous plants	
Betulaceae	Rhamnaceae
Alnus incana (L.) Moench	*Ceanothus sanguineus* Pursh
A. rubra Bong.	*C. velutinus* Dougl.
A. sinuata (Regel) Rydb.	
	Rosaceae
Elaeagnaceae	*Cercocarpus ledifolius* Nutt.
Shepherdia canadensis (L.) Nutt.	*Purshia tridentata* (Pursh) DC.
Leguminous plants	
Astragalus adsurgens Hook.	*Lupinus* spp.
A. agrestis Dougl.	*L. argenteus* Pursh
A. alpinus L.	*L. caudatus* Kell.
A. atropubescens Coult. & Fish.	*L. laxiflorus* Dougl.
A. canadensis L.	*L. lepidus* Dougl.
A. drummondii Hook.	*L. leucophyllus* Dougl.
A. kentrophyta Gray	*L. polyphyllus* Lindl.
A. miser Dougl.	*L. sericeus* Pursh
	L. wyethii Wats.
Hedysarum boreale Nutt.	
H. occidentalis Greene	*Oxytropis sericea* Nutt.
H. sulphurescens Rydb.	
	Thermopsis montana Nutt.
Lathyrus bijugatus White	
L. nevadensis Wats.	*Trifolium gymnocarpon* Nutt.
L. ochroleucus Hook.	*T. kingii* Wats.
L. pauciflorus Fern.	*T. latifolium* (Hook.) Greene
	T. longipes Nutt.
Lotus nevadensis (Wats.) Greene	*T. parryi* Gray
	Vicia americana Muhl.

[1]Information from Pfister and others (1977); Steele and others (1981, 1983); Cooper and others (1987).

Purshia was restricted to the drier sites, while *Alnus* was found in cooler, wetter stands. *Ceanothus* showed a scattered distribution in these late seral-climax stands, especially in Idaho. Ponderosa pine (*Pinus ponderosa* Laws.) and Douglas-fir (*Pseudotsuga menziesii* [Beissn.] Franco) sites had the greatest occurrence of N-fixing plants, which likely reflects their more open stand structure.

Although species of at least one nitrogen-fixing plant occurred in all but six of the 115 habitat types examined in Montana, Idaho, and Wyoming, these plants are not major understory components in most older Inland Northwest forests. Even when several N-fixing species are found on the same site, as often happens, the combined effect rarely averages more than 10 percent canopy coverage (Jurgensen and others 1979). However, some genera are so predominant on certain sites they have been designated the understory indicator for the habitat type: *Pinus flexilis/Cercocarpus ledifolius* (PIFL/CELE), *Pinus ponderosa/Purshia tridentata* (PIPO/PUTR), *Pseudotsuga menziesii/Cercocarpus ledifolius* (PSME/CELE), and *Abies lasiocarpa/Alnus sinuata* (ABLA/ALSI).

Early Seral Stands

Much less information is available on the distribution and frequency of N-fixing plants in early seral stands across the region. Most studies have been very site specific and usually detail early successional development after a disturbance, such as timber harvesting or fire. *Ceanothus* has received particular attention since it often becomes abundant after prescribed burns or wildfires. *Ceanothus* seed, which can remain viable in the soil for up to 200 years, requires a heat treatment to break dormancy (Noste and Bushey 1987).

Table 2—Occurrence of nitrogen-fixing plants in forest climax series of Montana[1]

N-fixing plant	*Pinus flexilis* (24)[2]	*Pinus ponderosa* (81)	*Pseudotsuga* (415)	*Picea* (100)	*Abies grandis* (30)	*Thuja* (42)	*Tsuga* (36)	*Abies lasiocarpa* (682)
	Percent of stands in which genus found							
Nonlegumes								
Alnus	0	0	23	17	17	12	14	15
Ceanothus	0	3	5	0	7	0	0	1
Purshia	4	23	7	0	0	0	0	0
Shepherdia	33	10	23	37	20	10	11	13
Legumes								
Astragalus	54	26	27	3	0	0	0	6
Hedysarum	21	5	6	12	7	0	0	5
Lathyrus	0	0	1	7	0	0	0	1
Lupinus	21	41	35	16	13	5	3	18
Oxytropis	4	4	1	0	0	0	0	0
Trifolium	0	0	1	2	0	0	0	2
Vicia	4	17	3	10	0	2	8	1

[1]Data are from late seral-climax forest stands (Pfister and others 1977).
[2]Total number of stands examined within each habitat series.

Table 3—Occurrence of nitrogen-fixing plants in forest climax series of Idaho and western Wyoming[1]

N-fixing plant	*Pinus flexilis* (34)[2]	*Pinus ponderosa* (99)	*Pseudotsuga* (709)	*Abies grandis* (381)	*Thuja* (269)	*Tsuga heterophylla* (153)	*Picea* (119)	*Tsuga mertensiana* (103)	*Abies lasiocarpa* (963)	*Pinus albicaulis* (53)
	Percent of stands in which genus found									
Nonlegumes										
Alnus	0	0	<1	3	3	1	3	3	3	0
Ceanothus	6	13	14	10	5	2	0	0	1	0
Cercocarpus	9	3	4	0	0	0	0	0	0	0
Purshia	9	36	8	<1	0	0	0	0	<1	6
Shepherdia	9	2	8	2	0	3	34	0	20	23
Legumes										
Astragalus	53	7	14	4	3	0	37	0	8	28
Hedysarum	20	5	2	0	<1	0	10	2	5	7
Lathyrus	0	15	3	11	7	<1	0	0	<1	0
Lotus	0	8	0	0	0	0	0	0	0	0
Lupinus	6	41	17	4	<1	1	3	2	18	23
Oxytropis	8	0	0	0	0	0	0	0	0	2
Thermopsis	0	2	<1	14	10	0	0	2	2	0
Trifolium	0	9	3	6	3	<1	3	0	2	2
Vicia	0	16	4	6	4	0	0	0	<1	0

Climax series.

[1]Data are from late seral and climax forest stands (Cooper and others 1987; Steele and others 1981, 1983).
[2]Total number of stands examined within each habitat series.

Reports of *Ceanothus* canopy coverage on burned sites in Idaho and Montana have ranged from less than 5 percent to over 80 percent (Arno and others 1985; Brown and DeByle 1989; Cholewa and Johnson 1983; Lyon 1971; Mueggler 1965; Noste 1985; Stickney 1980, 1986; Zamora 1975). Generally the hotter and more complete the burn, the greater the development of *Ceanothus* from buried seed (Noste and Bushey 1987; Orme and Leege 1976).

Other N-fixing plants would also be expected to increase after site disturbance, as part of the general increase in understory vegetation. However, few reports are available on the distribution of these plants in young stands of the Inland Northwest. Separate studies in northern Idaho cedar (*Thuja plicata* Donn ex D. Don)-hemlock (*Tsuga heterophylla* [Raf.] Sarg.) habitat types have reported *Alnus* to be either more common or more restricted after burning (Mueggler 1965; Stickney 1986; Wittinger and others 1977). In Montana, Stickney (1980) noted that *Alnus* frequency was reduced on subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.) sites after prescribed burning, but this species is commonly observed on road cuts or other scarified sites (Arno and others 1985). Growth and population density of *Purshia* are generally reduced after prescribed burns and wildfires in the Northern Rocky Mountains (Noste and Bushey 1987; Wagstaff 1980). *Shepherdia* showed a slight reduction in canopy coverage after a prescribed burn in a mixed aspen (*Populus tremuloides* Michx.)-conifer stand in western Wyoming (Brown and DeByle 1989). Both *Purshia* and *Shepherdia* are common mid-seral forest plants in central Idaho habitat types (Steele and Geier-Hayes 1987, 1989).

Little is known about legume distribution after forest disturbance. Some *Lupinus* species are fire survivors that are able to maintain themselves in the initial stages of plant succession after fire. Lyon and Stickney (1976) observed that species of *Lupinus* were abundant after wildfire in southwestern Montana. Canopy coverage of *Lupinus* was little changed after wildfire in a northern Idaho cedar-hemlock stand (Stickney 1986), or following prescribed burns in aspen or mixed aspen-conifer stands of eastern Idaho and northwestern Wyoming (Brown and DeByle 1989). *Astragalus miser* occurred in one of these Wyoming stands prior to burning, and was not present 4 years later. However, *Astragalus canadensis* became abundant following broadcast burning on grand fir (*Abies grandis* [Dougl. ex D. Don] Lindl.) sites in northern Idaho and northeastern Oregon and maintained a sizable population for at least 20 years (Zamora 1975). *Trifolium* was found on disturbed and burned sites in northern Idaho, but showed a rapid decline as the successional plant community developed (Mueggler 1965; Stickney 1986). Two species of *Hedysarum* responded strongly to burning and mechanical site preparation in Douglas-fir and subalpine fir habitat types in western Montana, while *Astragalus* and *Lupinus* did not (Arno and others 1985). Several nitrogen-fixing legumes (*Astragalus*, *Lupinus*, and *Thermopsis*) are important successional components of several central Idaho habitat types (Steele and Geier-Hayes 1987, 1989). The scattered and incomplete nature of these studies indicates that much more information is needed on the successional roles of N-fixing plants in this region.

Nitrogen Additions

Little information is available on the actual contribution of N-fixing plants to the N economy of Inland Northwest forests. The scattered distribution and low numbers of N-fixing plants in most late seral-climax stands suggests that annual N gains on such sites would be small. Fahey and others (1985) estimated that a *Lupinus argenteus*

density of 1,000 plants/ha in older, southeastern Wyoming lodgepole pine (*Pinus contorta* Dougl. ex Loud.) stands would add only 0.1 kg N/ha/yr. *Lupinus* growing in Utah aspen stands were reported to fix 0.6 kg N/ha/yr (Skujins and others 1987).

In contrast, much greater amounts of N could be added by N-fixing plants in successional forests. Studies on moist, highly productive sites in western Oregon and Washington have reported additions up to 100 kg of N/ha/yr from *Ceanothus* and *Alnus*, which became established on cut or burned sites (Conard and others 1985; Kimmins and others 1985). Much less information is available on the generally drier sites in the Inland Northwest, but indications are that N gains from N-fixing plants are lower. Youngberg and Wollum (1976) estimated N gains from *Ceanothus velutinus* (70 percent canopy cover) on a clearcut central Oregon ponderosa pine site to be 72 kg N/ha/yr over a 10-year period. A similar 11-year-old stand of *Ceanothus velutinus* (64 percent canopy cover) in northeastern Oregon was reported to fix 32 kg N/ha/yr (McNabb and others 1979). *Lupinus arcticus* (21,600 stems/ha) growing on a cutover lodgepole pine site in southern British Columbia was estimated to fix 2 kg N/ha/yr (Hendrickson and Burgess 1989). *Shepherdia* was also present on this site and added an additional 0.75 kg N/ha/yr.

The presence of N-fixing plants on a site in high numbers does not necessarily mean that significant amounts of N are being fixed. Dalton and Zobel (1977) estimated that less than 0.1 kg N/ha/yr was added to ponderosa and lodgepole pine stands in central Oregon by understories of *Purshia* (20 percent canopy cover). The amount of light reaching the soil surface and soil moisture levels during the growing season are critical factors in determining N-fixing plant activity (Sprent and Sprent 1990).

NONSYMBIOTIC NITROGEN FIXATION

The occurrence of N-fixing plants in most habitat types in the Inland Northwest should not obscure the fact that these plants are lacking or of low frequency on many sites (tables 2 and 3). On these sites, nonsymbiotic N-fixation becomes an important source of N to replace N lost from timber harvesting.

Nonsymbiotic N-fixation rates are strongly related to soil organic matter contents, being much higher in woody residue and surface organic layers than in mineral soil (table 4). Organic matter is required as an energy source for most N-fixing microorganisms (Jurgensen and Davey 1970), and has a high water-holding capacity. However, when these N-fixing rates were multiplied by soil weight/volume ratios to obtain the total amount of N fixed for each soil component, N gains in mineral soil also became important. This was especially evident on a cedar-hemlock site in Idaho, which had mineral soil N-fixation rates 500-600 percent greater than three sites in Montana (table 4). Favorable temperatures and moisture in the fertile northern Idaho ash cap soil encouraged the incorporation of surface organic matter into the mineral soil. Such higher organic matter levels would stimulate the activity of N-fixing bacteria (Granhall and Lindberg 1980).

Nitrogen Gains

The amounts of N fixed by nonsymbiotic N fixation in forest soils of the Inland Northwest generally reflect site productivity (table 4). Of the four old-growth stands examined, the highest N gain was found in a very productive, northern Idaho cedar-hemlock stand, and the lowest in a dry Douglas-fir stand in western Montana. These N-fixation differences were related to amounts of surface organic materials on each site, especially large woody residue (table 5). The greater the woody residue loadings on a site, the greater the N gains. On warm, dry sites, the accumulation of woody residue and other surface organic matter is reduced. Consequently, total N fixation on dry sites is lower than on wetter, more productive sites, while the proportion of N fixed in the mineral soil of these drier sites is greater.

The residue weights for the stands shown in table 4 are higher than regional averages for the Inland Northwest (Jurgensen and others 1987). Therefore, N fixation in woody residue on the majority of sites in this region would likely be lower than the results presented here. However, on many sites with heavy residue loadings, such as in overmature stands on moist sites or after a recent harvest, N gains could be much higher.

The results shown in tables 4 and 5 are from measurements taken during one year—1977. Longer term measurement of soil N-fixation rates has shown considerable annual variation (table 6). These N-fixation differences were related to yearly fluctuations in soil temperature and moisture. The highest N gains were obtained in a

Table 4—Nonsymbiotic nitrogen fixation in old-growth forests

Soil component	Subalpine fir (Montana)		Cedar-hemlock (Montana)		Douglas-fir (Montana)		Cedar-hemlock (Idaho)	
	ng N/d[1]	*g N*[2]	*ng N/d*	*g N*	*ng N/d*	*g N*	*ng N/d*	*g N*
Wood residue	21.0	515	15.6	230	18.6	159	35.2	1,428
Forest floor	32.5	328	15.7	192	12.1	101	14.0	88
Soil wood	26.3	250	7.0	91	9.1	95	19.1	178
Mineral soil	.7	379	.6	326	.6	442	4.0	1,197
Total		1,472		839		797		2,891

[1]Nanograms (10^{-9}) of N fixed/g of soil/day from June-October 1977 as measured by the acetylene reduction technique using a 3:1 ethylene to N conversion ratio.

[2]Total N fixed/ha over a 180-day period. Mineral soil sampled to a depth of 30 cm.

Table 5—Contribution of surface organic materials to soil nonsymbiotic nitrogen fixation in four old-growth forests

Soil component	Subalpine fir (Montana)		Cedar-hemlock (Montana)		Douglas-fir (Montana)		Cedar-hemlock (Idaho)	
	Mg[1]	*% N fix*[2]	*Mg*	*% N fix*	*Mg*	*% N fix*	*Mg*	*% N fix*
Wood residue	145.7	35	83.2	27	45.1	20	154.3	49
Forest floor	36.0	22	49.7	23	26.3	13	23.2	3
Soil wood	35.9	17	50.5	11	37.0	12	47.9	7
Mineral soil	—	26	—	39	—	55	—	41

[1]Dry weight (metric tonnes/ha) of organic material on top of mineral soil.
[2]Percentage of total N fixation shown in table 4.

Table 6—Annual fluctuations in soil nonsymbiotic nitrogen fixation on three old-growth sites in western Montana[1]

Year	Subalpine fir	Cedar-hemlock	Douglas-fir
	g N/ha/180d		
1976	1,246	1,817	1,496
1977	957	609	638
1978	348	309	346
1980	130	209	307
1981	1,097	985	832
Average	756	786	724

[1]Nitrogen fixed in the forest floor, soil wood, and the surface 30 cm of mineral soil as measured by the acetylene reduction technique.

cool, wet year (1976), while the lowest were measured in a warm, dry year (1980). Surprisingly, the average N fixation for these three stands over this 5-year period was quite similar. However, these values do not include N fixation in woody residue.

Many studies have tried to estimate the amounts of nonsymbiotic N fixation in forest soils of different timber types. Values in the literature range from <0.1 to 55 kg N/ha/yr (Boring and others 1988; Dawson 1983; Kimmins and others 1985), but most studies in the Western United States have reported N-fixation rates of <2 kg N/ha/yr (table 7). A direct comparison of these results is difficult due to the different soil layers tested and the various experimental conditions used to measure N fixation. Also, most of these studies were conducted for relatively short times (1 year or less), and as shown in table 6, annual fluctuations in N fixation can be considerable.

Harvesting Impacts

Nonsymbiotic N fixation is especially susceptible to harvesting impacts, since it is dependent on adequate organic matter supplies. The greater the amounts of organic matter removed or destroyed by timber harvesting and site treatments, the greater the possible reduction in N fixation. This effect was studied on a clearcut cedar-hemlock site in northern Idaho that had four harvest/site preparation treatments (table 8). The largest decrease in N fixation (63 percent) occurred after the prescribed burn, which removed 62 percent of the forest floor, soil wood, and woody residue. Slash removal by a bulldozer lowered N fixation by 48 percent, while clearcutting without any site preparation reduced N fixation by only 16 percent. In contrast, N fixation on the heavy residue treatment was 33 percent greater than in the uncut stand. Harvesting and site preparation destroyed much of the forest floor, except on the heavy slash treatment. However, decayed wood in the soil was much less disturbed by logging operations, and became a more important source of N fixation than the forest floor after harvesting. Soil wood is a major organic matter component in many Inland Northwest forest soils and generally retains more moisture than the forest floor during dry summer months (Page-Dumroese and others, these proceedings).

Woody residue was an important source of nonsymbiotic N fixation in the cedar-hemlock stand prior to cutting. Much of this material was removed from the site or destroyed by the prescribed burn and intensive harvesting treatments (table 5). This was especially evident on the tractor-piled treatment, where woody residue loadings were reduced by >90 percent. The low N fixation on both the intensive harvest and burn treatments reflected the low woody residue levels. In contrast, the amount of woody residue remaining after clearcutting was nearly the same as in the uncut stand, but the N fixed was 30 percent less. A similar pattern was seen in the heavy slash treatment, where woody residue loadings were 60 percent higher than in the uncut stand, but N fixation was nearly equal.

The contribution of woody residues to soil N fixation on this site depended on both the amount and type of woody material left after harvest. In the heavy slash treatment 45 percent of the woody residue was in the crumbly or solid rot stage, and 55 percent was undecayed. Solid rot residue is sound enough to withstand fragmentation during logging operations, while crumbly rot residue is easily destroyed (Benson and Schlieter 1980). Decayed wood of both rot types amounted to 59 percent of the total residue on the clearcut treatment. In contrast, 95 percent of the woody residue on the uncut site was large decaying logs, mostly in the crumbly rot stage. Many of these logs were destroyed during harvest and were not present on the cut treatments. They were replaced by smaller pieces of sound wood from harvested trees, which have much lower rates of N fixation (Jurgensen and others 1987).

Table 7—Nonsymbiotic nitrogen fixation in forests of the western United States

Forest type	Location	Source	N fixation	Reference[1]
			kg N/ha/yr	
Douglas-fir				
old growth	Oregon	Woody residue	1.0	1
	Oregon	Woody residue	1.4	2
	Montana	Woody residue, forest floor, mineral soil	0.8	4
various ages	Montana	Woody residue	0 - 1.0	3
23 yr old	Oregon	Forest floor	0.4 - 1.1	5
Subalpine fir				
old growth	Montana	Woody residue	0.7	6
various ages	Montana	Woody residue	0 - 1.7	3
	Montana	Woody residue, forest floor, mineral soil	1.5	4
Cedar-hemlock				
old growth	Idaho	Woody residue, forest floor, mineral soil	2.9	4
	Montana	Woody residue, forest floor, mineral soil	0.8	4
various ages	Idaho	Woody residue	0 - 4.3	3
	Montana	Woody residue	0 - 1.8	3
Mixed conifer				
various ages	Idaho	Living trees	<0.1 - 4.8	7
120 yr old	British Columbia	Woody residue, leaves, bark, forest floor, mineral soil	0.3	8
Lodgepole pine 80 yr old	Wyoming	Woody residue	<0.2	9
Aspen—mature	Utah	Forest floor, mineral soil	0.5	10

[1](1)Sollins and others 1987; (2) Silvester and others 1982; (3) Jurgensen and others 1987; (4) this paper—table 4; (5) Heath and others 1988; (6) Larsen and others 1978; (7) Harvey and others 1989; (8) Cushon and Feller 1989; (9) Fahey and others 1985; (10) Skujins and others 1987.

Table 8—Nonsymbiotic nitrogen fixation on a cedar-hemlock site in northern Idaho after timber harvesting and woody residue removal[1]

Soil component	Residue treatment: None		Residue treatment: Prescribed burn		Residue treatment: Intensive removal		Residue treatment: Heavy residue		Uncut	
	Mg[2]	*gN*[3]	*Mg*	*gN*	*Mg*	*gN*	*Mg*	*gN*	*Mg*	*gN*
Wood residue	146.0	984	57.9	177	10.6	111	249.8	1,483	154.3	1,428
Forest floor	16.7	110	5.5	26	13.3	73	34.5	326	23.2	88
Soil wood	50.9	109	22.4	47	51.6	195	50.3	430	47.9	178
Mineral soil	—	1,218	—	826	—	1,125	—	1,608	—	1,197
Total	213.6	2,421	85.8	1,076	75.5	1,504	334.6	3,847	225.4	2,891

[1]Site was clearcut to a 12.7-cm diameter top. Residue treatments: none—residue left; prescribed burned—broadcast burned in the fall; intensive removal—residue removed by blading with a crawler tractor; heavy residue—residue removed by blading was added to residue left on another area.

[2]Dry weight (metric tonnes/ha) of organic material on top of mineral soil.

[3]Total N fixed/ha over a 180-day period—1977. Mineral soil sampled to a depth of 30 cm.

REPLACEMENT OF NITROGEN LOSSES

While nonsymbiotic N fixation was generally lower after harvesting a northern Idaho cedar-hemlock stand (table 8), the actual reduction was quite small (1-2 kg/ha/yr). The question is whether such small losses in N fixation are important to maintaining long-term site productivity. Using soil organic matter weights before and after harvest (table 8), the N content for these materials (Page-Dumroese and others, these proceedings), and assuming N is lost in proportion to weight losses, 440 kg of N were estimated to have been lost from this cedar-hemlock site by clearcutting and prescribed burning. This does not include any N losses that may have occurred in the mineral soil. Another 200-250 kg N were likely removed from the site in bolewood (Prescott and others 1989). Using an N-fixation gain of 1.08 kg/ha/yr (table 8), and 1.5 kg N/ha/yr added in precipitation, these N losses would be replaced in 250 to 270 years. Such a slow return to original soil N levels could have a considerable impact on subsequent stand growth.

This calculation was based on the assumption that inputs from N fixation will not change as the next stand develops. Nitrogen fixation should increase in the forest floor as this layer increases in thickness during stand development. Nitrogen fixation in the mineral soil was reduced by burning (table 8), but would likely increase as the forest floor becomes thicker. However, these N gains would be at least partially offset by a decrease of N fixation in woody residue. Residue weights would likely decrease with time, as wood decomposition rates are usually greater than residue inputs from young, fast-growing stands. It is only after 100 to 150 years that woody residue begins to increase as mature trees die (Harmon and others 1986; Spies and others 1988). A similar situation would probably occur for the soil wood.

Other sources of nonsymbiotic N fixation not accounted for in these calculations could also add appreciable amounts of N to the soil, and reduce the time required for site recovery. Tree stumps, cull trees left for snags, and large dead roots can add up to 1 kg/ha/yr after harvesting (Granhall and Lindberg 1980; Harvey and others 1989). Increased development of N-fixing algae and lichens on the soil surface after harvest could also add N, but this would likely be small due to generally dry conditions throughout the summer. Nitrogen-fixing bacteria may also be active in the rhizospheres of developing tree seedlings and shrubs (Amaranthus and others 1990).

Nitrogen-fixing plants are present in many cedar-hemlock habitat types in the Inland Northwest (tables 2 and 3). If any of these plants became established on the harvested site, soil N losses could be replaced more rapidly than by nonsymbiotic N fixation alone. Assuming symbiotic N fixation on this cedar-hemlock site could vary from a low of 3 kg N/ha/yr (Hendrickson and Burgess 1989) to a high of 72 kg N/ha/yr (Youngberg and Wollum 1976), the recovery time to preharvest soil N levels would range from 10 to 125 years.

MANAGEMENT IMPLICATIONS

Nitrogen-fixing plants have the potential to add significant amounts of N to forest sites in the Inland Northwest. Of the many N-fixing plants present in this region, *Ceanothus* seems most amenable to management. Fire, as part of postharvest site treatments or stand underburning, would favor *Ceanothus* development in many habitat types. *Alnus*, *Shepherdia*, *Lupinus*, and *Astragalus* also have management possibilities on many sites, but much more information is needed on the response of these genera to stand disturbance in a wide range of habitat types.

While the N added by N-fixing plants could be important in replacing N losses from harvesting or fire, these plants are serious competition for tree seedlings on many sites (McDonald and Fiddler 1989; Petersen and others 1988; Stewart and others 1984). In the short term, reduced stand growth from such plant competiton often far outweighs the benefits from added soil N. Youngberg and Wollum (1976) recommended using *Ceanothus* for soil N enrichment during initial seedling development, followed by herbicide treatment to release the trees. Whether such a chemical treatment would be economically or environmentally possible in the Inland Northwest is questionable. A better alternative would be to create a mosaic of microsite conditions using fire or mechanical scarification, which allows the development of both N-fixing plants and adequate tree regeneration (Geier-Hayes 1987), or to artificially establish N-fixing plants after harvest (Everett and others, these proceedings).

Another option would be to develop silvicultural systems to minimize competing vegetation, but leave as much organic matter on the soil surface as possible. This would favor the activity of nonsymbiotic N-fixing bacteria. While annual N inputs from these bacteria are small, such N gains over the life of the stand can be appreciable. Maintaining soil N levels is critical to continued productivity of Inland Northwest forests. These forests must be managed to minimize N losses from timber harvesting activities, and to encourage N inputs from biological sources.

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ECTOMYCORRHIZAL ACTIVITY AND CONIFER GROWTH INTERACTIONS IN WESTERN-MONTANE FOREST SOILS

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ABSTRACT

The distribution of ectomycorrhizal activities in forest soils and their potential interactions with conifer growth in a variety of ecological / age / treatment situations are described. In old-growth ecosystems ectomycorrhizal roots are concentrated in organic soil horizons. In planted situations positive correlations between early seedling growth and ectomycorrhizal activity were consistent the first year and most often associated with high organic matter soils and competition thereafter. Potential effects of a variety of management actions on soil-related ectomycorrhizal inputs to productivity, and the opportunities for ectomycorrhizal technology to improve early performance in regenerated forest stands are discussed.

INTRODUCTION

Widespread recognition of the critical role of symbiont activities, particularly ectomycorrhizae, in survival, growth, and long-term productivity of vegetative components in wildland ecosystems has provided great impetus for research on the effects of management actions on these microbial activities (Amaranthus and others 1989; Harvey and others 1979a, 1987; Jurgensen and others 1979; Perry and others 1987, 1989). Soil conditions are particularly important in governing the ability of ectomycorrhizal fungi and their respective hosts to initiate the association (Amaranthus and others 1989; Bjorkman 1970; Page-Dumroese and others 1990; Perry and others 1987; Slankis 1974) and, as we shall see, are also likely to directly affect the functioning of the association. Thus, forestry operations that substantially alter soil conditions, particularly surface horizons, are likely to alter both initiation and function of ectomycorrhizal associations and, subsequently, growth, competitive ability (St. John and Coleman 1983), and succession in the regenerated forest.

Extensive examinations of the distribution of forest tree feeder roots (Grier and others 1981; McKay and Malcolm 1988; Vogt and others 1981, 1982, 1983) and ectomycorrhizae in forested ecosystems (Fogel and Hunt 1983; Harvey and others 1976b, 1979b, 1987; St. John and Coleman 1983; Vogt and others 1982) have shown they tend to occur in shallow, relatively fertile organic horizons where they are highly subject to disturbance (fig. 1). Frequently, decayed woody deposits in and on the soil are important to ectomycorrhizal associations, particularly during dry seasons and on dry sites (Harvey and others 1978, 1979b). Decaying logs and soil wood have been recognized as unique components of ecosystems with important ties to ectomycorrhizal activity and potentially to productivity in both inland (Harvey and others 1976b, 1979a, 1987, 1989; Larsen and others 1980) and Pacific forests (Harmon and others 1986; Maser and others 1985, 1988).

Potential for interruption in the development or destruction of surface organic horizons and resulting effects on ectomycorrhizal activities lead to a concern that inappropriate use of prescribed fire and intensive utilization on the relatively infertile soils and harsh forests of the inland West could pose a substantial danger to future productivity (Harvey and others 1976a, 1979a; Jurgensen and others 1979). Similarly, there has been concern that site preparation could also disrupt surface organic horizons and related

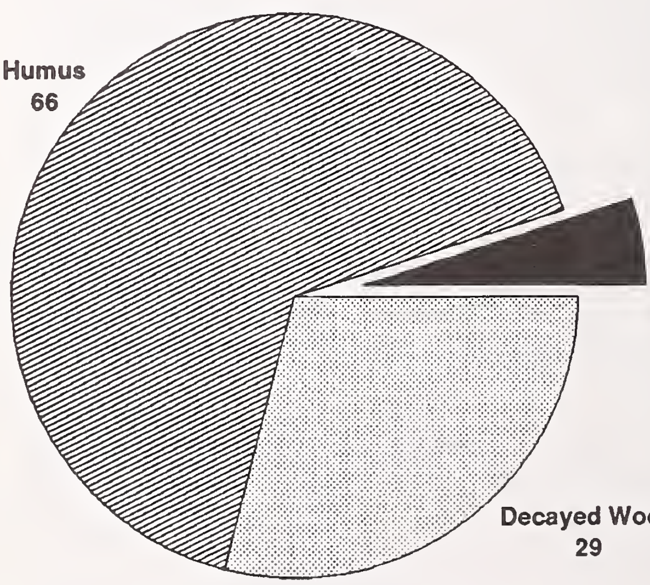

Figure 1—Distribution (percentage) of ectomycorrhizae among soil horizons in western Montana (from Harvey and others 1976b).

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productivity and regeneration potential of inland western forest soils (Graham and others 1989b; Harvey 1982).

The strong differential relationship between soil organic and mineral horizons and ectomycorrhizal activities under differing conditions and on differing sites (Alvarez and others 1979; Harvey and others 1978, 1979b) suggested the possibility that function of the ectomycorrhizal association, as well as initiation of the relationship, might be controlled by soil microsite conditions. The cost in energy to trees for maintaining this association, both at the stand (Fogel and Hunt 1983; Perry and others 1987) and individual tree/seedling level (Marshall and Perry 1987; Miller and others 1989), can be very high. However, tree growth may or may not be directly improved by the presence of ectomycorrhizal roots. Although in many instances presence of ectomycorrhizal activity determines both survival and growth potential for conifers (Amaranthus and Perry 1987; Perry and others 1987; Trappe and Strand 1969), in other instances there has not been a demonstrably positive effect (Krop and others 1985; Molina 1982; Shaw and others 1987; Sidle and Shaw 1987). Thus, specific information is not available regarding exactly when and where ectomycorrhizal relationships are most critical to tree survival and growth and when and where they may or may not have an immediate, critical role in the development of forest stands.

As a result of this lack of or conflicting information, we developed an experiment where site preparation treatments were applied to native inland western forest soils to provide an analysis of the effects of soil microsite conditions on both numbers of ectomycorrhizae and their impact on tree growth. Some early results of this experiment are presented here.

OBJECTIVES

The objectives for the experiment were to: (1) create a situation where seedlings of two important conifer species with widely differing strategies for site adaptation (sharp vs. gradual genetic clines) could be grown in disturbed, native, forest soils while being subjected to different levels of stress as a consequence of both environment (site preparation treatment) and competing vegetation, and (2) assess ectomycorrhizal development on the planted seedlings and relate that development to performance. Since organic matter, moisture, and nutrition were known to affect ectomycorrhizal development, and stress was likely to impact ectomycorrhizal function, they were incorporated into the basic experimental design.

METHODS

The experiment was conducted on two different sites located in the Priest River Experimental Forest in northern Idaho. One was located on a low bench (715 m) with relatively poor soil (Typic Xerochrept), subject to extremes of temperature and moisture, with well-developed grass, sedge, and herb competition (harsh site). This site was typical of the grand fir/snowberry (*Abies grandis/Symphoricarpos albus*) habitat type in northern Idaho. The other was located on a midslope (1,456 m) not subjected to such extremes of temperature and moisture, with a relatively good soil (Typic Cryothent), and lacking the well-developed competitive community (moderate site). This site was typical of the western hemlock/clintonia (*Tsuga heterophylla/Clintonia uniflora*) habitat type in northern Idaho. See Cooper and others (1987) for more details on these habitat types.

Superimposed onto these sites were four site preparation treatments: (1) mounded soil beds where soil organic matter volume was increased and competing vegetation left in place; (2) mounded soil beds with competing vegetation removed, manually in year 1, with herbicide (isopropylamine salt of glyphosate, 1.68 kg/ha) applied annually to nonconifer vegetation only in years 2 and 3; (3) a scalp where competing vegetation, organic horizons, and mineral topsoil were removed to a depth of 5 cm; and (4) a minimally scarified area that was undisturbed after harvesting. Two randomized complete block experiments were established on each of the sites; at the low-elevation site there were four replications and on the midslope site three replications of each treatment.

Each site/treatment combination was planted with 240 1-year-old, containerized Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) or western white pine (*Pinus monticola* Dougl. ex D. Don) seedlings in April 1983. All seedlings were grown from locally collected seed and were essentially devoid of ectomycorrhizae at the time of planting. Sixteen randomly selected, live seedlings were subsequently excavated from each treatment at approximately 6-week intervals, starting in May and continuing through the growing season. Four sampling times were included: (1) spring, (2) early summer, (3) late summer, and (4) fall for each of the first 3 years after planting. These sampling times were equivalent to early May, mid-June, early August, and mid-September, respectively. All ectomycorrhizal root tips were counted by type (Harvey and others 1976b, 1979b); root and top dry weights (24 h at 105 °C) were recorded for each seedling. Soil conditions around each of the seedlings were fully characterized, including physical and chemical parameters (Page-Dumroese and others 1986, 1989). Survival rates by treatment were also recorded (Graham and others 1989a).

Analysis of variance (Steele and Torrie 1960) was used to test the effects of site, tree species, site preparation, soil, and sample time (season/year) on numbers of ectomycorrhizae. Scheffe's multiple range test was used to detect significant differences among means (Mize and Schultz 1985). Pearson's correlation (SASS) was used to determine relationships between numbers of ectomycorrhizal short roots and seedling growth characteristics.

RESULTS

Seedling Weight—Douglas-fir (DF) and western white pine (WWP) responded to site preparation treatments in a similar manner on both sites (figs. 2 and 3). As expected, the heaviest trees occurred on both sites in the mounded treatment that had the competing vegetation removed. Survival was severely limited ($P < 0.05$) only on the harsh site in mounds with the intense competition left intact (36 percent WWP, 54 percent DF survival after 3 years, compared to at least 76 percent survival for all other site, species, and treatment combinations). Beyond that, however,

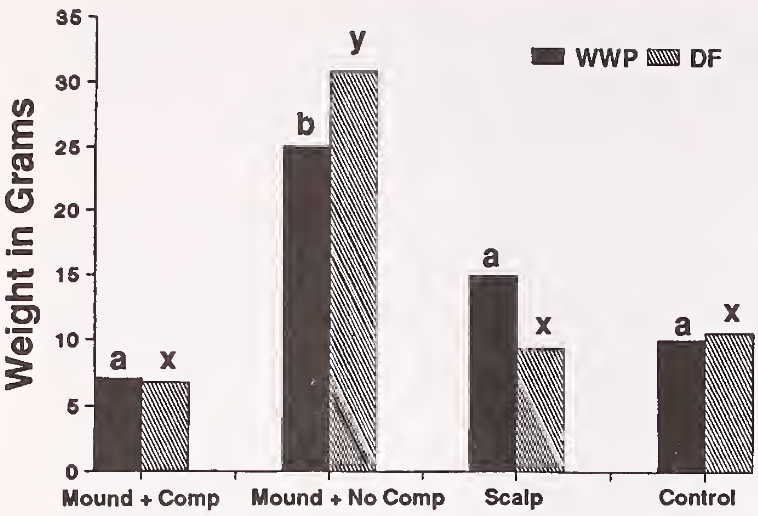

Figure 2—Total seedling biomass after 3 years (harsh site). Differing letters indicate significant differences ($P < 0.05$) between treatments.

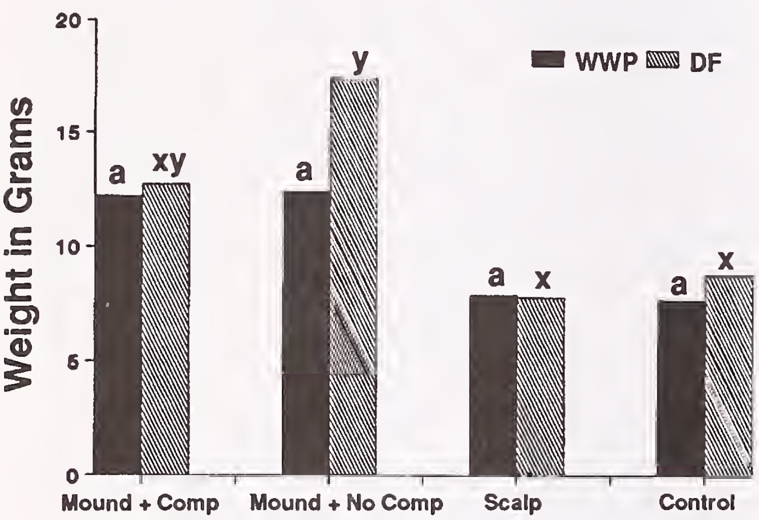

Figure 3—Total seedling biomass after 3 years (moderate site). Differing letters indicate significant differences ($P < 0.05$) between treatments.

seedling response was not always as expected. For example, leaving competing vegetation in place on the harsh site in the mounded treatment completely negated the positive effects of reduced bulk density, increased nutrients, organic matter, and moisture (fig. 2). Also, despite the physical removal of competition from both the harsh (fig. 2) and moderate (fig. 3) site scalps, performance in those areas was no better than for the minimal disturbance treatments where competition was left in place.

In general, the pattern of performance that emerged with both species was strong positive responses to the presence and quantity of organic soil components and a strong negative response to competition. Where positive and negative elements were combined, they tended to offset one another.

Ectomycorrhizal Development—Initiation of ectomycorrhizal associations on the root systems of first-year seedlings was rapid on WWP seedlings and relatively slow on DF. Table 1 provides an example. A full complement

Table 1—Effect of season on ectomycorrhizae (first year, harsh site)

		Ectomycorrhizal tips	
Site	Season	No competition	Scalp
WWP	Spring	[1]4.8a	6.1x
	Early summer	48.5ab	88.1y
	Late summer	81.7b	93.2y
	Fall	100.3b	74.5xy
DF	Spring	0.0a	0.3x
	Early summer	.1a	1.0x
	Late summer	7.5a	21.4x
	Fall	25.2b	45.1y

[1]Different letters indicate significant differences ($P < 0.05$) between seasons, within species, 16 seedling samples.

of ectomycorrhizal short roots (no further significant differences, $P = 0.05$) was established on the WWP seedlings after only 12 weeks. In contrast, increases in ectomycorrhizae occurred on DF seedlings throughout the full 24-week sampling period in the first season.

Treatment conditions imposed strong controls on the number of ectomycorrhizal short roots (ESR's) supported by individual seedlings of both species (figs. 4 and 5). In general, species responded similarly; significantly higher numbers of ESR's occurred on seedlings in the scalped treatments compared to other treatments. Additionally, the warmer of the two sites (harsh site) generally supported more ESR's per seedling than the middle-elevation (moderate) site for most treatments.

Season (sampling time) also had a strong effect on numbers of ESR's occurring on the seedlings. This effect was most pronounced with WWP seedlings in the mounded treatment with no competing vegetation, and it varied with site (table 2).

Higher numbers of ESR's occurred on WWP than DF seedlings throughout the treatments on either site (figs. 4 and 5, table 3). This relationship was consistent and did not depend on size of root system or seedling top.

In summary, variables within the experiment that influenced numbers of ESR's per seedling included: (1) first year age, (2) treatment, (3) site, and (4) species. No significant differences in ectomycorrhizal morphology were noted when comparing the treatments and variables described here.

Relationship Between Ectomycorrhizae and Seedling Weight—Not surprisingly, these same variables also affected correlations between the numbers of ESR's supported on seedling root systems and performance of those seedlings (total weight, root weight, or top weight).

Starting with age, tables 4 and 5 show first-year weight was positively correlated, at varying significance levels, with numbers of ESR's across all treatments on both sites. This was the most consistently positive relationship between ectomycorrhizae and growth characteristics that was observed throughout the experiment.

After the first year, correlations were not uniformly positive for all treatments (tables 4 and 5). For example, the relationship between growth and ESR's was often positive only for the mounded treatments with heavy competition.

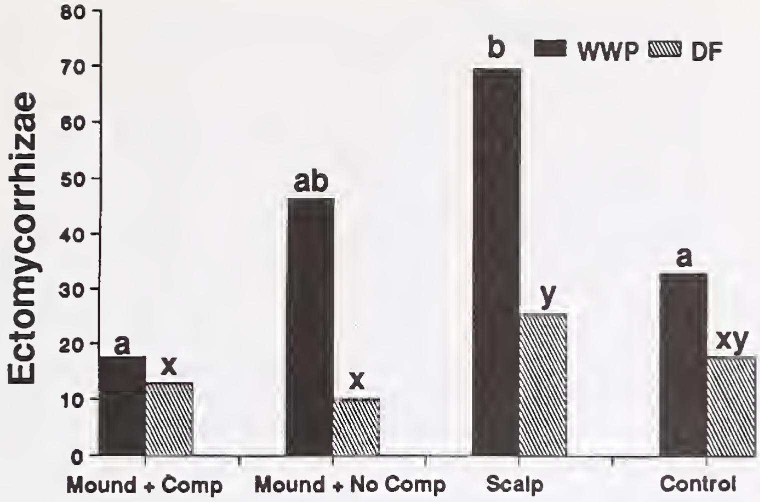

Figure 4—Effect of treatment on numbers of ectomycorrhizal tips per seedling after 3 years (harsh site). Differing letters indicate significant differences ($P < 0.05$) between treatments.

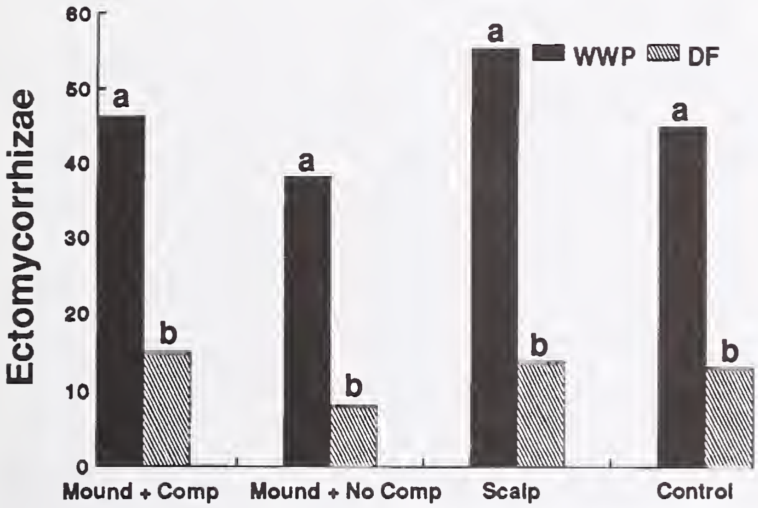

Figure 5—Effect of treatment on numbers of ectomycorrhizal tips per seedling after 3 years (moderate site). Differences among treatments were not significant, differences between species were significant ($P < 0.05$) on all treatments.

Table 2—Effect of season and site on ectomycorrhizal root tips of WWP (second year, mound, no competition)

	Ectomycorrhizal tips	
Season	Harsh site	Moderate site
Spring	[1]52.3a	61.9x
Early summer	23.2b	43.9x
Late summer	42.8a	18.6y
Fall	52.2a	15.7y

[1]Different letters indicate significant differences ($P < 0.05$) between seasons, within site, 16 seedling samples.

In the presence of heavy competition, positive correlations between ESR's and seedling weight remained relatively consistent, particularly with WWP.

Despite the relatively high numbers of ectomycorrhizal short roots on seedlings of both species in the scalped treatments, on either site with either species, relationships with growth were usually neutral (not significant) in years 2 and 3 (tables 4 and 5). In general, patterns of response between sites were similar but year-to-year variations were noted. For example, the mound treatment (with competition) for DF on the moderate site in year 2 showed a negative rather than a positive correlation (table 5).

Comparing the two species between sites showed strong differential reactions (table 6). For example, when competition and organic matter were high, ectomycorrhizae were positively correlated with seedling weight of both species only on the harsh site. There was a strong negative correlation between ectomycorrhizae and seedling weight of WWP on the moderate site.

Seasonal data within year 3 on WWP showed a significant seasonal effect. Both the early and late samples had neutral correlations with growth, but during the midsummer drought there was a positive correlation between numbers of ESR's and seedling weight (table 7).

Table 3—Effect of species on ectomycorrhizae and seedling weight (grams) after 3 years

		No competition		Scalp	
Species	Site	Ectomycorrhizal tips	Weight	Ectomycorrhizal tips	Weight
			Grams		*Grams*
WWP	Harsh	[1]60.1a	25.0a	98.8x	14.9x
DF		14.6b	30.8a	56.4y	9.4y
WWP	Moderate	55.1a	12.4a	65.4x	7.9x
DF		14.8a	17.4b	21.9y	7.8x

[1]Different letters indicate significant differences ($P < 0.05$) between species at each location, 16 seedling samples.

Table 4—Treatment and year effects on correlations between ectomycorrhizal tips and seedling weight (grams)

Harsh site (WWP)	Year 1	2	3
Mound + competition	+[1]	++	+
Mound, no competition	+++	+	– – –
Scalp	+	N	N
Control	++	+	N

[1]Sign indicates correlation with growth (+$P < 0.1$, ++$P < 0.05$, +++$P < 0.01$, – – –$P < -0.01$, N indicates neutral effect), 65 seedling samples.

Table 5—Treatment and year effects on correlations between ectomycorrhizal tips and seedling weight (grams)

Moderate site (Douglas-fir)	Year 1	2	3
Mound + competition	+++[1]	– – –	+++
Mound, no competition	+++	– – –	N
Scalp	+++	N	N
Control	+++	N	N

[1]Sign indicates correlation with growth (+$P < 0.1$, ++$P < 0.05$, +++$P < 0.01$, – – –$P < -0.01$, N indicates neutral effect), 65 seedling samples.

Table 6—Site and species effects on correlations between ectomycorrhizal tips and seedling weight (grams)

Site (mound + competition) year 3	Ectomycorrhizal tips Douglas-fir	Western white pine
Harsh	[1]22.2+	14.8+
Moderate	22.8+++	38.3–

[1]Sign indicates correlation with growth (+$P < 0.1$, +++$P < 0.01$, –$P < -0.1$), 63 seedling samples.

Table 7—Seasonal effects on correlations between ectomycorrhizal tips and seedling weight (grams)

Season	Ectomycorrhizal tips year 3, scalp, WWP
Spring	[1]132.8N
Early summer	54.9++
Late summer	117.2++
Fall	58.7N

[1]Sign indicates positive correlation with growth (++ $P < 0.05$, N indicates neutral effect), 26 seedling samples.

Table 8—Species effects on correlations between ectomycorrhizal tips and seedling growth (weight/age)

Treatment (all years included)	Ectomycorrhizal tips Western white pine	Douglas-fir
Mound + competition	[1]73.3+	38.3– –
Mound + no competition	43.1N	21.9– –
Scalp	64.7–	40.4– –
Control	40.5N	15.8– –

[1]Sign indicates correlation with growth (+P 0.1, –$P < -0.1$, – –$P < -0.05$, N indicates neutral effect), 500 seedling samples.

Combining data for all 3 years from both sites, then comparing species reactions to treatments, showed strong responses between seedling species and individual treatments (table 8). Thus, the same variables that controlled numbers of ESR's per seedling (development) also affected their correlations with growth (function). These variables controlled not only the strength of correlations, but also whether they were positive, negative, or neutral.

DISCUSSION

We believe these results make it quite evident that the nature of the soil treatment, the site, the season, the species, and time all affect both the ability of seedlings to form ectomycorrhizal associations and the ability of those associations to contribute to seedling growth. Thus, the question regarding where and when ectomycorrhizae provide a favorable return in terms of seedling growth can only be answered when these variables are known. These results are similar to those obtained for endomycorrhizal associations on alpine grasses growing in wildland ecosystems (Fitter 1986).

Perhaps the only exception to this generalization was first-year performance of WWP and DF seedlings growing on these northern Idaho sites. During that year ectomycorrhizae were beneficial to both species for all treatments on both sites (tables 4 and 5). Early establishment of ectomycorrhizal associations appears critical to good performance, even on these relatively moderate sites. This relationship would likely have been even more critical if harsher sites had been used (Amaranthus and Perry 1987).

The relative slowness of DF seedlings to establish a good complement of ESR's during the first season suggests that outplanting seedlings with well-developed ectomycorrhizal associations could improve first-year performance of this species. Amaranthus and Castellano (in preparation) have found significant increases in survival and growth rates of Douglas-fir seedlings inoculated with spores of the ectomycorrhizal fungus *Rhizopogon vinicolor* on a variety of sites in the Pacific Northwest. However, WWP seedlings responded to the native inoculum on these sites so rapidly that it seems unlikely, except on extremely harsh sites or degraded soils, that routine efforts to produce appropriate ESR's in the nursery could be beneficial. These results also suggest that seral species, with their ability to form ESR's quickly, may be a better choice to reforest many highly disturbed sites with poor soils in northern Idaho than would DF. Other species that fill climax roles in these ecosystems may also be slow to initiate ectomycorrhizal associations.

If, however, seedlings are to be planted on sites where competitive ability would be a major factor in early survival and growth, results of this study suggest that strong mycorrhization of outplanting stock at the nursery would likely be beneficial, even to WWP. Additionally, the ability to positively affect growth in highly competitive situations strongly suggests that effective ectomycorrhizal development may be a key process, not only during early establishment and growth, but also during later stages of succession and, ultimately, in the highly competitive, advanced stages of forest development.

The inability of strong ESR development on both WWP and DF seedlings in scalp treatments to contribute significantly to increased seedling weight indicates that factors

limiting growth could not be improved substantially by the presence of active symbionts. It is notable, however, that even with the much higher numbers of ESR's on the smaller seedlings than with other treatments, the correlations between ESR's and growth were not negative as might have been expected. Thus, even with the limited photosynthetic capacity of small seedlings and the large attendant carbon drain brought about by supporting high ESR counts, growth returns to the seedlings were at least sufficient to offset the high relative cost. This, in turn, suggests that the scalp treatment was either lacking in factors needed for growth of the seedlings or it inhibited or delayed the ESR's from acquiring them.

Seasonal relationships between numbers of ESR's and their correlation with growth during midsummer suggest that ectomycorrhizal activities may be particularly important to conifer performance during the intermittent, climatic droughts typical of western-montane forest ecosystems.

APPLICATION PERSPECTIVES

Although it is evident from this and other studies that ectomycorrhizal associations can exact a cost to conifer seedlings that is high when compared to the direct returns (increased growth) provided by the association, significant benefits are most often provided in the most critical ecological circumstances: first-year establishment, during drought, and in highly competitive situations. Thus, maintaining ectomycorrhizae on conifer root systems can be viewed as the ecological equivalent of an insurance policy. In many cases the premiums are high when times are good, but in times of crises, benefits may be critical to survival, growth, and competitive ability. Additionally, there are likely to be significant, indirect soil benefits resulting from support of the microbial community in and around conifer root systems (see paper by Molina and Amaranthus, this proceedings).

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Speakers answered questions from the audience after their presentations. Following are the questions and answers on this topic:

Q.—What is the function of decaying roots in maintaining soil productivity? Do roots compensate for excessive organic matter removal from the soil surface?

A.—Decaying roots have important positive functions the same as any other buried woody materials, with the added advantage that they are somewhat protected from loss due to fire. In some situations decaying roots could compensate for loss of surface wood; however, in many inland ecosystems the volume of root material would be well short of the woody debris requirement and it might not replace forest floor horizons adequately because it is buried.

Q.—Are ectomycorrhizae important on dry ponderosa pine sites?

A.—Yes, they are fully as important as on any other site, perhaps more so because of the relative harshness of dry ecosystems.

Q.—Why was establishment of the ectomycorrhizal complement of Douglas-fir so slow compared to that of western white pine?

A.—We hypothesize that this is at least in part a genetically programmed trait. We assume that rapid formation of ectomycorrhizae is especially critical to species like the various pines that usually fulfill pioneer and early seral roles in these ecosystems. Thus, they would normally be faced with disturbed, possibly degraded soils and harsh environments.

Q.—Have you evaluated qualitative as well as quantitative aspects of ectomycorrhizal behavior in inland ecosystems?

A.—Yes. We have noted some differences in ectomycorrhizal types in various circumstances; for example, in general, the harsher the ecosystem or site the more types we are likely to see. However, the variation has usually been so high that specific differences are not supportable statistically.

ORGANIC MATTER DECOMPOSITION IN WESTERN UNITED STATES FORESTS

Robert L. Edmonds

ABSTRACT

Understanding decomposition processes and the influence of forest management practices on them is crucial to maintaining long-term productivity in western forests. This paper discusses: (1) organic matter accumulations in western forests, including coarse woody debris (CWD), (2) organic matter decomposition rates, including the effects of clearcutting, (3) physical, chemical, and biological factors influencing decomposition rates, and (4) nitrogen dynamics in decomposing substrates. Decomposition rates are much higher in coastal forests (k = 0.27-0.44 yr^{-1} for Douglas-fir needles) than in inland forests (k = 0.05-0.14 yr^{-1} for pine needles). Decomposition rates for woody substrates are one to two orders of magnitude slower depending on their size. Needle decomposition rates are increased by clearcutting. Nitrogen release from a substrate is related to its decomposition rate and N may be immobilized for a long time in CWD.

INTRODUCTION

Decomposition is the process whereby litter on the soil surface and belowground roots are broken down to smaller particles (Swift and others 1979). It releases soluble forms of nutrients that are available for plant uptake and provides soil organic matter (Waring and Schlesinger 1985). Understanding decomposition processes and the influence of forest management practices on them is crucial to maintaining the long-term productivity of western forests. Organic matter decomposition also contributes CO_2 to the atmosphere, thus influencing global warming.

Fungi and bacteria are the dominant decomposers in coniferous forests (Richards 1987). Small animals, such as mites, fragment fine litter and enhance microbial decomposition. Earthworms, although important in the decomposition process in deciduous forests, are not thought to play a major role in coniferous forests, but they are present (Richards 1987). Insects, especially termites, ants, bark beetles, and wood borers, play a very important role in the decomposition of woody litter by fragmenting wood and introducing fungal decomposers (Harmon and others 1986).

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Simple sugars decompose completely to CO_2 and water, but decomposition of the complicated organic substrates in forest ecosystems is not complete. Hard-to-decompose or recalcitrant substances accumulate in the soil as humus, which comprises the soil organic matter so important in maintaining forest productivity. Soil organic matter maintains soil structure, improves soil water balance, and is a long-term source of site nutrients. It is particularly vulnerable to loss through improper forest practices and is important in preventing compaction and erosion.

There are many sources of organic matter in western forests, and one of these is coarse woody debris (CWD), such as logs and snags. Determining the importance of CWD in western forests has been the focus of much research in recent years (Harmon and others 1986; Harvey and others 1979, 1981; Larsen and others 1980; Maser and others 1988), and maintaining CWD in western forests is one of the major components of "new" forestry (Eubanks 1989; Franklin 1989). Coarse woody debris provides: (1) plant habitat through nurse logs, (2) moisture and nutrients for fine roots and mycorrhizae, (3) habitat for animals and birds, (4) pools in streams for fish habitat, (5) sites for nitrogen (N) fixation, and (6) a long-term source of soil organic matter. It also protects against erosion by improving slope and stream stability, maintains species diversity, and helps in maintaining long-term site productivity. Many of these roles change as CWD decomposes.

The objectives of this paper are to: (1) present data on organic matter accumulation in western forests, (2) discuss organic matter decomposition rates in western forests including the effects of clearcutting on decomposition rates, (3) discuss the factors influencing organic matter decomposition rates, and (4) examine N dynamics in decomposing substrates.

ORGANIC MATTER ACCUMULATIONS

The sources of soil organic matter in western forests are fine litterfall (needles, leaves, insect frass, etc.), fine woody litterfall (twigs, branches, cones), coarse woody debris (logs, snags, and stumps), roots (fine and coarse), and soil organisms. The importance of fine roots in contributing organic matter to the soil has only recently been demonstrated. Fine root turnover can be equal to needle litter inputs and in some ecosystems, for example, Pacific silver fir, fine root inputs can be three times aboveground fine litter inputs (Vogt and others 1986).

The amount of dead organic matter accumulating in an ecosystem is the balance among litter and root inputs, decomposition, and the effects of fire. Organic matter accumulations in western U.S. forests are among the highest in the World's forests (Cole and Rapp 1981). There is considerable variability, however, in organic matter accumulations in this zone, with highest accumulations in productive wetter and cooler coastal areas and lowest accumulations in less productive hotter and drier inland areas (table 1). A large proportion of this accumulation is in CWD and soil organic matter. Highest CWD accumulations occur in old-growth forests in the western Olympic Mountains (> 500 Mg/ha) (Agee and Huff 1987) with lesser amounts in old-growth forests in the western Oregon and Washington Cascades (averaging 60-220 Mg/ha) (Spies and others 1988).

Fire is the major disturbance in western forests and 600-1,000 Mg/ha of CWD can be found immediately after a catastrophic wildfire in old-growth forests (Spies and others 1988). This CWD will decompose with time, and new accumulations will begin at about age 50 years. Lowest amounts of CWD (< 100 Mg/ha) tend to occur 100 to 200 years after the fire, after which the amount of CWD will increase again. Windstorms can also add large amounts of coarse woody debris in areas close to the coast.

Forest management has changed organic matter accumulations, particularly with respect to CWD (Harmon and others 1990; Spies and others 1988). Spies and others (1988) suggest forest management activities greatly reduce the amount of CWD below minimums typically encountered under natural ecosystem dynamics. For most of the managed rotation CWD biomass is < 30 Mg/ha. This is supported by data from second rotation forests in the Puget Sound area where surface CWD biomass is only around 30 Mg/ha (R. L. Edmonds, unpublished data). In third rotation forests CWD biomass is further reduced to 10 Mg/ha. Stumps left after harvest only account for about 3 Mg/ha in third-rotation forests, but coarse woody roots contribute significantly more (around 30 Mg/ha) (R. L. Edmonds, unpublished data). This loss of a long-term source of organic matter is of considerable concern.

Soil organic matter is also likely to be reduced with forest management (Harmon and others 1990), although few studies have been conducted to examine this. The optimum level of organic matter in soils to maintain productivity in western forests is not known. However, organic matter removal on poor Douglas-fir sites tended to have a greater effect on reducing productivity than on good sites (Bigger 1988). The forest floor, roots, and fine woody litter become increasingly important contributors to soil organic matter as the intensity of forest management increases and the contribution of large woody litter decreases.

Table 1—Organic accumulations (Mg/ha) in mature western U.S. forests

Location	Coastal (Douglas-fir/hemlock)	Inland (pine)
Litter and humus	[1,2]16-57	[3,4]5-33
Fine roots	[5]5-13	
Coarse woody debris		
logs	[2,6,7,8]42-500	[6,4]1-104
snags	[6]25-105	[6]2-41
coarse roots	[2]97-193(live)	[9]26-56(live)
Soil	[1]79-776	[4]490-591

[1]Johnson and others (1982).
[2]Grier and Logan (1977).
[3]Fahey (1983).
[4]Fahey and others (1985).
[5]Vogt and others (1986).
[6]Harmon and others (1986).
[7]Spies and others (1988).
[8]Agee and Huff (1987).
[9]Pearson and others (1984).

DETERMINATION OF DECOMPOSITION RATES

Organic matter decomposition rates are usually determined by examining substrate dry weight or mass loss, changes in specific gravity, or carbon dioxide evolution from a substrate. Substrate mass loss with time is typically used to determine decomposition rates of fine litter (needles, leaves, twigs, cones, bark, small branches, etc.), whereas specific gravity change is usually used for determining CWD decomposition rates. Although the specific gravity of large boles can be determined relatively easily, it is more difficult to determine how long a bole has been on the ground unless the bole is a result of a known blowdown. Typically, logs that have been on the ground for a long time are aged by examining adjacent living trees for scars and aging the scars using an increment borer (Sollins and others 1987). The age of trees growing on downed logs can also be determined. Both fragmentation and respiration losses have to be taken into account for boles and snags.

Carbon dioxide evolution is not typically used for determining decomposition rates of specific substrates, because it is such a dynamic measure and it is hard to integrate over a long time period. It is also difficult to separate substrate respiration from root respiration, and forest-floor CO_2 evolution is not always well related to decomposition rates (Vogt and others 1980).

Typically, decomposition rates are expressed as k values or fractional loss rates rather than just changes in mass or specific gravity with time. The unit of k is typically yr^{-1} or per year. A simple negative exponential curve can be used to express decomposition and the k value can be determined from the equation $X/X_o = e^{-kt}$, where X_o = initial dry mass or specific gravity and X = mass or specific gravity at time t (years) (Olson 1963). Typically, k values using this equation decrease with time because the rate of decomposition is more rapid early in the process and slows with time (Edmonds 1984; Yavitt and Fahey 1982). More complex models have also been developed (for example, Bunnell and Tait 1974; Means and others 1982; Melillo and others 1989). The k values for CWD commonly have a fragmentation component (k_{frag}) and a respiration component (k_{resp}) or (k_{min}) (Harmon and others 1986).

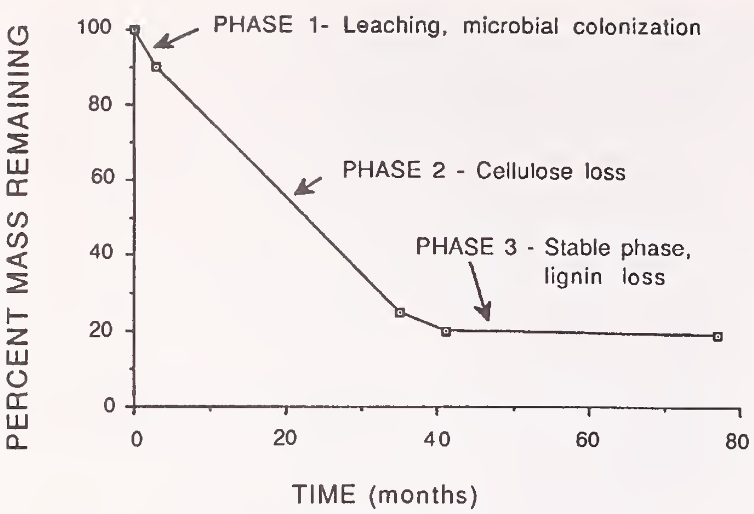

Figure 1—Phases of decomposition in red pine needles in Massachusetts. From Melillo and others (1989).

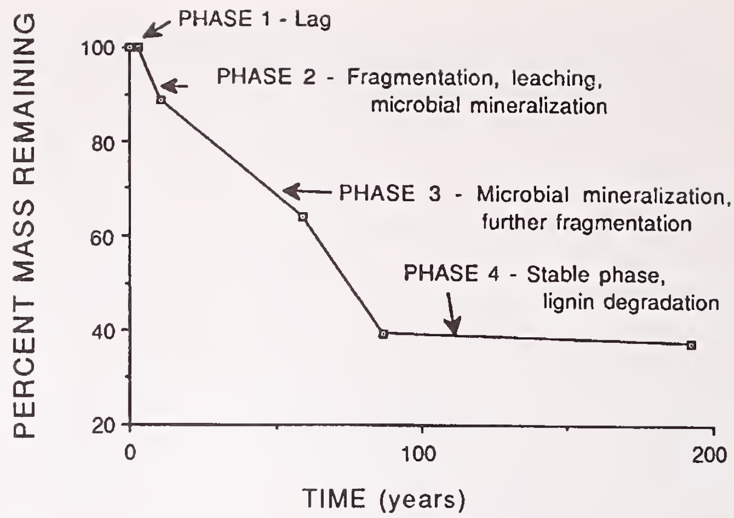

Figure 2—Phases of decomposition of Douglas-fir logs in western Oregon and Washington. Data from Sollins and others (1987).

THE DECOMPOSITION PROCESS

Decomposition of fine litter, such as needles or leaves, occurs in three phases as shown in figure 1 for red pine in the eastern U.S. (Melillo and others 1989). In the first phase, during the first few months, the labile or fast fraction (sugars and starch, etc.) is lost by rapid microbial assimilation or leaching. Harmon and others (1990) noted that leached litter in coastal Washington decomposed slower than unleached litter. The second phase is dominated by loss of structural or slow carbon, which is primarily cell wall polymers such as cellulose. The third phase is the stable or metastable phase in which there is a very slow decrease in mass. This phase is dominated by lignin decomposition.

Mellillo and others (1989) have examined decomposition along a decay continuum from plant litter to soil organic matter and feel that litter mass loss can best be modeled using a two-phase model: an initial phase of constant mass loss and a very slow loss dominated by a degradation of "lignocellulose" (acid-soluble sugars plus acid-insoluble C compounds). As the decaying litter enters the second phase, the ratio of lignin to lignin plus cellulose (the lignocellulose index-LCI, or the fraction of lignin in the lignocellulose complex) approaches 0.7. Beyond this the LCI increases only slowly throughout the decay continuum indicating that acid insoluble materials (lignin) dominate decay in the later stages. For red pine needles the second phase began when the organic matter remaining was about 20 percent; it ranged between 15 and 30 percent for other litter materials (Melillo and others 1989).

Woody litter goes through a similar process but has at least one additional stage as shown in figure 2. Before any weight loss or change in specific gravity occurs there is a lag phase (phase 1), which is usually related to the size of the substrate (larger woody substrates usually have a longer lag time). In phase 2, logs begin to weather and fragment. Leaching losses and microbial activity occur. After the period of active fragmentation there is a period of rapid microbial mineralization (phase 3).

Table 2—Decay classification for Douglas-fir boles (from Maser and others 1988)

Characteristics of fallen trees	Decay class I	II	III	IV	V
Bark	Intact	Intact	Trace	Absent	Absent
Twigs	Present	Absent	Absent	Absent	Absent
Texture	Intact	Intact to partly soft	Hard, large pieces	Small, soft, blocky pieces	Soft and powdery
Shape	Round	Round	Round	Round to oval	Oval
Color of wood	Original color	Original color	Original color	Light brown to reddish brown	Red brown to dark brown
Portion of tree on ground	Tree elevated on support points	Tree elevated on support points; sagging slightly	Tree sagging near ground	All of tree on ground	All of tree on ground
Invading roots	None	None	In sapwood	In heartwood	In heartwood

is followed by the stable phase, which is dominated by lignin decomposition (phase 4). Most coniferous logs at this stage consist of a mass of crumbly brown cubical rot. Maser and others (1988) have described five visual decay classes for Douglas-fir logs as shown in table 2 involving the presence or absence of bark, twigs and roots, texture, shape, color, etc. Note that roots do not begin to invade logs until they reach decay class III.

DECOMPOSITION RATES IN WESTERN FORESTS

The study of organic matter decomposition in western forests is of relatively recent origin. Few studies were conducted before the 1970's, and most of these involved examining bole deterioration, particularly after windstorms (Shea and Johnson 1962) or insect epidemics (Wright and Harvey 1967). They did not determine actual decomposition rates.

Interest in studying organic matter decomposition rates in western forests from an ecological viewpoint increased after the initiation of the Coniferous Forest Biome Program of the International Biological Program (IBP) in the 1970's (Edmonds 1982). A number of studies were initiated in western Oregon and Washington at that time, mostly focusing on needle decomposition (for example, Edmonds 1979, 1980, 1984; Fogel and Cromack 1977). The IBP also stimulated interest in studying decomposition rates of coarse woody debris, and a number of studies were conducted starting in the late 1970's (Graham and Cromack 1982; Grier 1978; Harmon and others 1986; Sollins 1982; Sollins and others 1987; Spies and others 1988).

Decomposition rates of needles, roots, branches, bark, and wood in western forests, including inland sites, are summarized in table 3. Douglas-fir was taken to be a representative species for coastal sites, while data for pines are presented for inland sites. There are differences, however, in decomposition rates among different species occurring in a local area. For example, in the Washington Cascades k values for low-elevation red alder, Douglas-fir, western hemlock, and high-elevation Pacific silver fir are 0.45, 0.44, 0.38, and 0.30 yr^{-1}, respectively, based on 2 years of decomposition (Edmonds 1980).

Overall decomposition rates for needles are much higher in coastal Douglas-fir (k = 0.27-0.44 yr^{-1}) than in pine types in California (k = 0.05 yr^{-1}), Arizona (k = 0.14 y^{-1}), and Wyoming (k = 0.14 yr^{-1}), due to moister conditions in coastal forests. This is also illustrated in figure 3, which shows isolines of first-year needle and leaf decomposition rates in the U.S. The predicted rates are higher than actual rates because first-year rates are usually very rapid. Note that decomposition rates in the West are in general considerably slower than those in the eastern U.S., especially the southeastern U.S. This largely reflects different climatic regimes. It is considerably warmer and wetter in the southeastern U.S. than it is in the West and summers tend to be dry in the West even in coastal areas. The factors controlling decomposition rates are discussed in more detail in the following section.

Table 3—Typical average decomposition rates of needles, branches, roots, bark, and wood in western U.S. forests

	Coastal (Douglas-fir)		Inland (pines)	
Substrate	k (yr^{-1})[1]	Time to 95% decay (yr)[2]	k (yr^{-1})	Time to 95% decay (yr)
Needles	0.44 (WA)[3]	7	0.05 (CA)[4]	60
	0.27 (OR)[5]	11	0.14 (AZ)[6]	21
			0.14 (WY)[7]	21
Branches	0.06 (WA)[8]	121		
Roots				
fine	0.18 (OR)[9]	17	0.05 (WY)[10]	60
coarse			0.027(WY)[10]	111
Bark	0.03 (OR)[5]	100		
Wood (diam., cm)				
24 (log)	0.026 (WA)[11]	115		
37 (log)	0.050 (WA)[11]	60		
>65 (log)	0.006 (WA, OR)[12]	500		
	0.029 (WA, OR)[13]	103		

[1]Based on single exponential model of Olson (1963).
[2]3/k (Olson 1963).
[3]Edmonds (1980a).
[4]Stohlgren (1988a).
[5]Fogel and Cromack (1977).
[6]Klemmedson and others (1985).
[7]Yavitt and Fahey (1986).
[8]Edmonds (1987).
[9]Fogel and Hunt (1979).
[10]Yavitt and Fahey (1982).
[11]Edmonds and Eglitis (1989).
[12]Harmon and others (1986).
[13]Spies and others (1988) (includes both fragmentation and respiration).

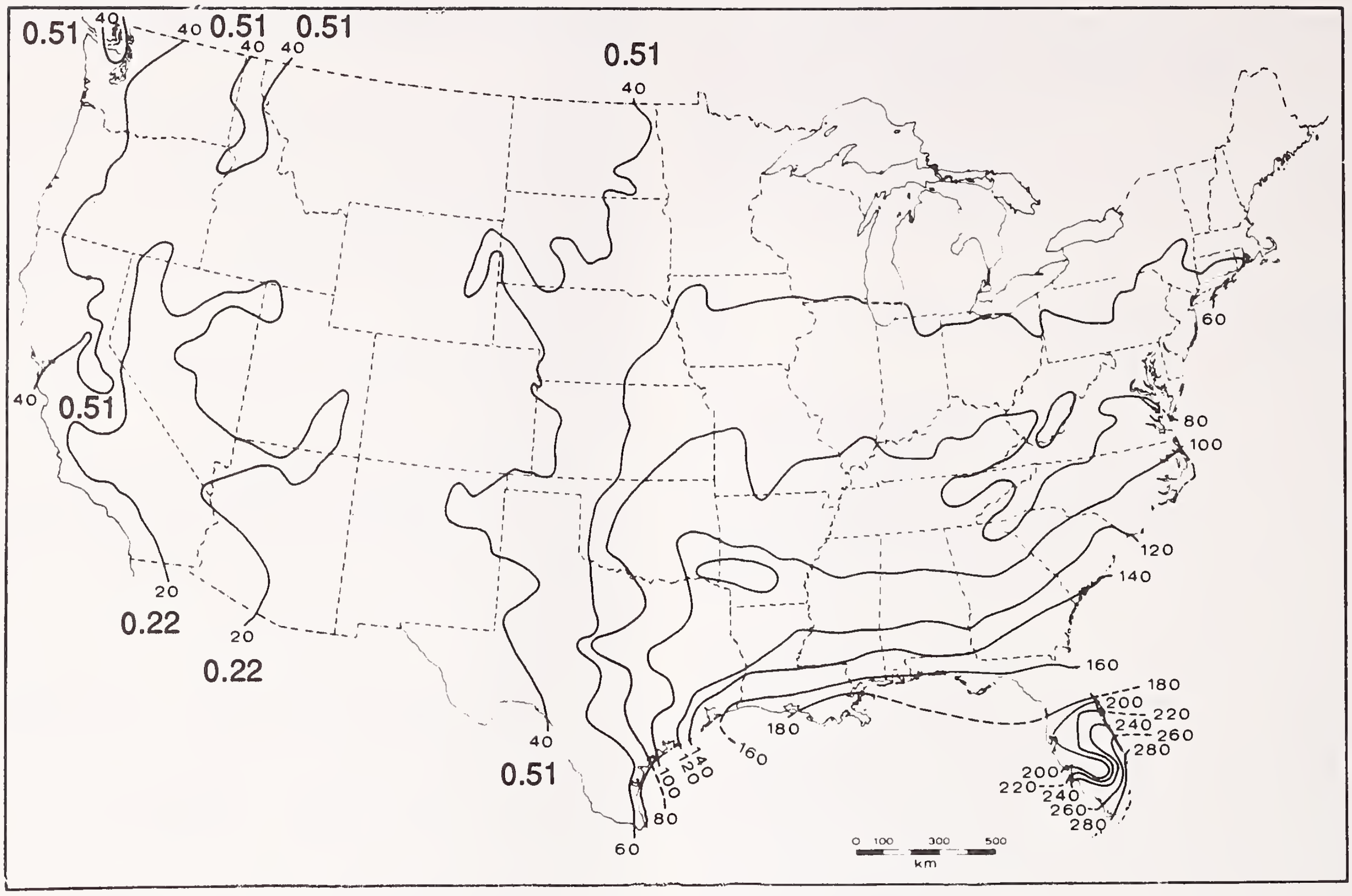

Figure 3—Isopleths of percent needle and leaf litter produced annually that will decompose in the first year. *k* values (yr^{-1}) are shown for the western U.S. Adapted from Meentemeyer (1978a).

are discussed in more detail in the following section. Decomposition rates also change with stand age, with the fastest rate occurring near the time of canopy closure (Edmonds 1979).

There are few data on fine root decomposition rates in the West, but the data that do exist indicate that rates are slightly slower or similar to those for foliage (table 3). Berg (1984) also noted this for Scots pine fine roots. Coarse woody roots in Wyoming decomposed more slowly ($k = 0.027\ yr^{-1}$) than fine roots ($k = 0.05\ yr^{-1}$). This *k* value for coarse roots, however, was calculated after 12-15 years of decomposition. When calculated after 80-110 years it decreased to $0.008\ yr^{-1}$ (Yavitt and Fahey 1982). Coarse root wood appears to decay very slowly in the later stages of decomposition. Yavitt and Fahey (1982) noted there was still a large amount of root wood at their sites in Wyoming 100 years after tree death. Bark also decomposes slowly, and Douglas-fir bark decomposes at a similar rate ($k = 0.03\ yr^{-1}$) to small logs (table 3).

Considerably more decomposition data exist for surface woody litter. Wood decomposes one to two orders of magnitude slower than needles depending on size (table 3). Branches decompose at a faster rate than logs, and small logs decompose at a faster rate than large logs (table 3). Edmonds and Eglitis (1989), in an exception to this rule, however, noted that small Douglas-fir logs (average diameter 24 cm) decomposed at a slower rate than medium-sized logs (average diameter 37 cm). Small logs were not as easily attacked by wood-boring insects, which apparently spread wood-rotting fungi. This was only a 10-year study, however, during the initial period of decomposition.

Large Douglas-fir logs in coastal forests may exist on the forest floor for more than 500 years based on a simple *k* value (table 3). Spies and others (1988), however, suggested that using k_{resp} values to calculate the longevity of logs may be misleading. They found much higher *k* values ($0.029\ yr^{-1}$) when fragmentation losses were included, resulting in a much shorter longevity (closer to 100 years). They feel that this is a more realistic rate based on CWD accumulations in Cascade Mountain forests. Nevertheless, it is apparent that Douglas-fir logs exist for an extended period on the forest floor, performing important ecological roles as they decompose. A typical old-growth Douglas-fir tree may live for 500 years, but it continues to function for 100 years or more after death.

Logs of different tree species have slightly different decomposition rates. Sollins and others (1987) determined *k* values (0.009, 0.016, and 0.010 yr^{-1}) for western redcedar, western hemlock, and Douglas-fir, respectively. However, they also noted that no recognizable western redcedar or hemlock logs greater than 100 years old could be found, yet Douglas-fir logs almost 200 years old were recognizable. Thus, there seems to be a problem in using *k* values for determining exact log life.

Table 4—Decomposition rates (k values, yr^{-1}) of needles, twigs, and branches in clearcuts in western U.S. forests

Material	Closed canopy	Clearcut	Forest edge
Douglas-fir needles (WA)[1,2]	0.44	0.39-0.53	0.29-0.51
Ponderosa pine needles (AZ)[3]	0.14	0.34	
Douglas-fir twigs and branches (WA)[4,5]	0.06-0.14	0.01-0.04	
Ponderosa pine (WA)[5]		0.01	

[1]Edmonds (1980).
[2]Edmonds and Bigger (1984).
[3]Klemmedson and others (1985).
[4]Edmonds and others (1986).
[5]Erickson and others (1985).

There appears to be a lack of data for log decomposition rates for inland sites in the West (Harmon and others 1986), but the impression is that logs may not last as long in inland sites as they do in coastal sites. At inland sites logs are smaller, tend to be easily colonized by insects, and of course tend to burn more easily in an environment where the fire frequency is higher. Standing dead trees or snags may have high fragmentation decomposition rates at inland sites (k_{frag} = 0.073-0.318 yr^{-1}) resulting from physical exposure and the action of insects and cavity-nesting birds (Harmon and others 1986). Snags in general tend to have higher decomposition rates than logs. For example, for Douglas-fir snags k_{frag} = 0.014-0.354 yr^{-1} and k_{min} = 0.003-0.027 yr^{-1}. For Douglas-fir boles k_{frag} = 0.008 yr^{-1} and k_{min} = 0.004-0.007 yr^{-1}. Note that k values for fragmentation are higher than k values for mineralization.

The influence of forest management on litter decomposition rates currently holds considerable interest, especially decomposition rates in clearcuts. Table 4 shows that decomposition rates of Douglas-fir in Washington and pine needles in Arizona are higher in clearcuts than in closed-canopy stands. Entry and others (1987) also found higher decomposition rates for needles in clearcuts in Montana. Decomposition rates may also be higher at forest edges than beneath closed canopy stands (table 4). This has important implications with respect to nutrient availability and tree growth. In contrast to needles, twigs and branches in clearcuts seem to have a slower rate of decomposition than those in closed canopy stands (table 4), perhaps because they dry out rapidly in clearcuts.

FACTORS INFLUENCING DECOMPOSITION RATES

Many interacting factors are involved in the decomposition of organic substrates. A list of physical, chemical, and biological factors influencing decomposition is shown in table 5; these factors are discussed below.

Physical Factors

Temperature and moisture play important roles in the decomposition process. Decomposition is generally faster in cool, moist areas and slower in hot, dry areas, and this is generally the case in the western U.S. as illustrated by the data in table 3 and figure 1. There is tremendous variation in temperature and moisture in the West because of the mountainous terrain. Elevation and slope factors can change temperature and moisture conditions over relatively short distances. Forest management effects such as clearcutting also can dramatically change temperature and moisture conditions. Moisture conditions in clearcuts in the early summer may be more conducive for decomposition in clearcuts because of lower evapotranspiration. Meentemeyer (1978b) found that decomposition rates were negatively related to actual evapotranspiration. Temperatures are also warmer in clearcuts.

Substrate size or surface-to-volume ratio is also important. Fine litter has a high surface-to-volume ratio and is more easily colonized by microbes than woody substrates. Very large diameter logs decompose slowly (table 3), reflecting their small surface-to-volume ratios. Decomposition of woody substrates, however, is not strictly determined by surface-to-volume ratio. Insects are extremely important in wood decomposition, and certain groups of insects, such as wood borers, may not be as effective in attacking small logs as larger logs, thus influencing decomposition rates.

Table 5—Physical, chemical, and biological factors influencing organic matter decomposition rates

1. **Physical factors**
   a. Temperature
   b. Moisture
   c. Actual evapotranspiration
   d. Size—surface-to-volume ratios
   e. Substrate location—buried, forest edges, clearcuts, etc.—reflects temperature and moisture
   f. Oxygen (aeration) and CO_2
   g. Fire
2. **Chemical factors—substrate chemistry**
   a. Nitrogen concentration
   b. C/N ratio
   c. Lignin concentration
   d. Lignin/N ratio
   e. Lignocellulose index
   f. Extractives
   g. pH
3. **Biological factors**
   a. Microbes—fungi, bacteria
   b. Soil animals—especially earthworms and insects
   c. Plant species—reflects substrate chemistry

Substrate location is also very important. Buried wood, for example, decomposes much faster than wood on the surface of clearcuts with aerial wood decomposing even more slowly (Edmonds and others 1986). This largely reflects different temperature and moisture conditions. Decomposition rates are also lower in locations with low aeration and high CO_2 (Griffin 1972), for example, in very wet soils. Decomposition proceeds more slowly in anaerobic environments because fungi are excluded and only bacteria are present.

Fire is not usually considered as an agent of decomposition. It does, however, oxidize organic matter and could be considered a decomposing agent in hot, dry ecosystems where microbial decomposition is slow. It may also influence the ability of substrates to microbially decompose after a fire. Most fires completely oxidize fine litter, but charred woody substrates may remain. Little information is available on decomposition rates of charred logs, but observation would suggest that charred logs decompose at a slower rate than noncharred logs. Certainly charcoal can remain in the soil for centuries after a fire. Entry and others (1987) found that lignin and cellulose decomposition was slower in clearcut and burned sites in Montana than in unburned clearcuts.

Chemical Factors

Considerable research has been conducted on the influence of substrate chemistry on decomposition rates. Generally, substrates with high N concentrations and low C:N ratios decompose at the fastest rate. However, the relationship is more complex than this, and as a result attention has focused on other factors such as substrate structural chemistry. Plant substrates are made up of noncell-wall components (waxes, resins, phenolics, sugars) and cell-wall structural chemicals (cellulose and lignin). Slowly decomposing substrates are generally high in lignin (Fogel and Cromack 1977). Melillo and others (1982) suggested that both lignin and N (lignin:N ratio) may be a better predictor of decomposition rates than lignin or N alone. The lignocellulose index also appears to be related to decomposition rates (Melillo and others 1989).

The concentration of extractives (for example, phenolics) in substrates is also important in determining decomposition rates (Swift and others 1979). Extractives are materials that can be removed by solvents such as hot water, ethanol, or benzene. Red alder leaves in coastal Washington decompose slower than predicted by the lignin:N ratio (Harmon and others 1990) perhaps because of their high extractives content.

Extractives are particularly important in controlling decomposition rates of woody substrates. They typically give color to heartwood. Thus species with dark-colored heartwood, such as redwood, Douglas-fir, and western redcedar, are typically more decay resistant than white wood species such as hemlocks and true firs.

Acidity or pH can also influence decomposition rates. At low pH, microbes are inhibited, especially bacteria (Richards 1987).

Biological Factors

Biological factors are extremely important in the decomposition process. Without the presence of microbes and their enzymes, organic matter decomposition would be very slow (Ugolini and Edmonds 1983). Fungi are the dominant agents of decomposition in aerobic environments. They possess enzymes such as cellulase to more efficiently break down complex substrates, and they can tolerate more acidic environments than bacteria. Organic acids are produced during the decomposition process, which tend to result in an acid soil environment, particularly in cold environments.

Mycorrhizae are important in western forests with respect to nutrient uptake by plants (Amaranthus and others 1989; Maser and others 1988). However, it has been suggested by workers in New Zealand that the presence of mycorrhizae may inhibit litter-decomposing fungi (Gadgil and Gadgil 1975). This, however, has not as yet been demonstrated in western U.S. forests. Griffiths and others (1990), however, have demonstrated complex interactions among Douglas-fir trees, ectomycorrhizal fungi, and the microbial community in ectomycorrhizal mats in the soil.

Soil animals such as earthworms are important in decomposition of most deciduous leaf litter (Richards 1987). Although native earthworms are present in acidic western coniferous ecosystems, they are generally not thought to play an important role in the decomposition process. Spiers and others (1986), however, argue that the role of indigenous earthworms in decomposition and nutrient cycling in Northwest coastal ecosystems may have been underestimated. They observed as many as 200 worms m^2 in some ecosystems. Other dominant soil animals in coniferous forests are the enchytraeid worms, mites, and insects such as Collembola, which tend only to fragment organic matter or graze on fungi and bacteria (Edmonds 1980b).

Although insects may be not be very important in the decomposition of fine litter, they play an extremely important role in wood decomposition, particularly in the early phases (Carpenter and others 1988; Edmonds and Eglitis 1989). They fragment woody substrates and introduce fungal spores and mycelia. Carpenter ants, bark beetles, wood borers, and termites are the dominant insects involved.

Finally, in terms of biological factors, plant species is important (Daubenmire and Prusso 1963). Each species has different substrate chemistries in needles, leaves, roots, branches, and boles, and this strongly influences their decomposition rates (Edmonds 1980a; Sollins and others 1987).

NITROGEN DYNAMICS IN DECOMPOSING SUBSTRATES

Nitrogen is the major growth-limiting nutrient element in the Northwest (Edmonds and others 1989), and its availability is largely determined by its rate of release

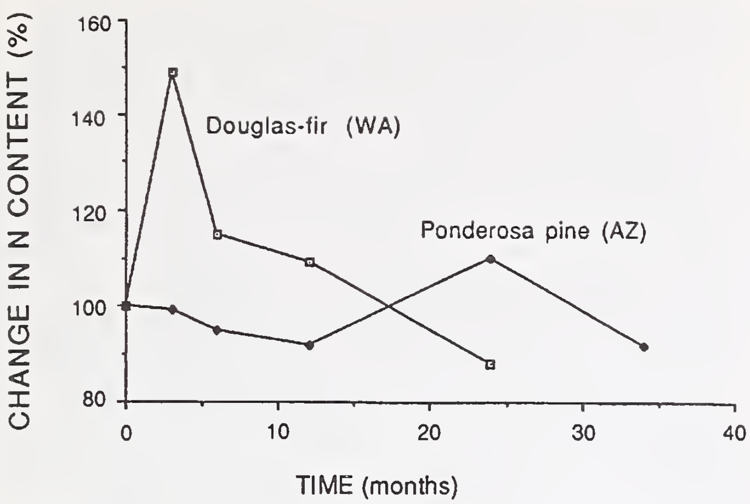

Figure 4—Percent of original N mass in decomposing Douglas-fir needles in Washington (Edmonds 1980a) and ponderosa pine needles in Arizona (Klemmedson and others 1985) in relation to time.

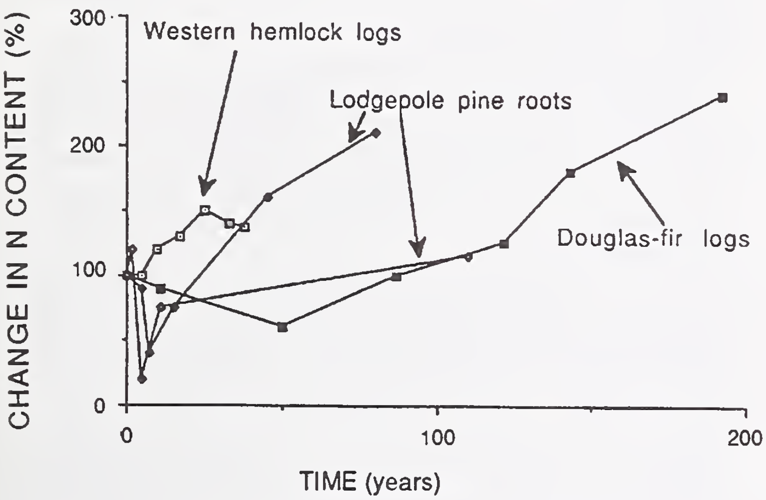

Figure 5—Percent of original N mass in decomposing western hemlock logs in coastal Oregon (Grier 1978), Douglas-fir logs in the Washington and Oregon Cascades (Sollins and others 1987) and lodgepole pine woody roots in Wyoming (Yavitt and Fahey 1982) in relation to time.

from decomposing organic matter. Fresh litter falling to the forest floor is decomposed by fungi and bacteria, which typically immobilize any N in the decomposing substrate for a period of time before releasing it to the soil where it becomes available for plant uptake. Figure 4 illustrates the fate of N with time in decomposing Douglas-fir needles in Washington (Edmonds 1980a) and ponderosa pine needles in Arizona (Klemmedson and others 1985). Nitrogen was immobilized in Douglas-fir needles only in the first 3 months, while it appeared to be immobilized in ponderosa pine needles for 24 months. Stohlgren (1988b) also noted a long period of immobilization in needle litter in some Sierra Nevada ecosystems in California.

The length of this period of immobilization depends on the concentration of N in the initial substrate. Substrates high in N, such as red alder leaves, release N almost immediately on reaching the forest floor (Edmonds 1980a), while logs that have a low N concentration may immobilize N for as long as 25 years (Grier 1978) or even 190 years (Sollins and others 1987) (fig. 5). Yavitt and Fahey (1982) found that the N content of decaying woody roots was still increasing after more than 100 years of decomposition (fig. 5).

The C:N ratio of decomposing substrates continuously declines with time and with depth in the soil profile. For example, it may be 50:1 in freshly fallen Douglas-fir needles (Edmonds 1980a), below 30:1 in the forest floor, and less than 20:1 in the soil (Edmonds and Hsiang 1987). A critical C:N ratio has been suggested as controlling N release in decomposing substrates. Lutz and Chandler (1946) suggested that N mineralization should occur at C:N ratios between 20 and 30:1 with immobilization occurring at ratios greater than this. This appears to be the case for decomposing leaf and needle litter (Edmonds 1980, 1984), but it does not appear to be the case for woody substrates. Critical C:N ratios for N mineralization appear to be > 100:1 for twigs and branches and > 300:1 in large logs (Edmonds 1987; Sollins and others 1987). Thus the critical C:N ratio for N release is not constant but increases as the substrate decomposition rate decreases. There also appears to be no N release until lignin decomposition starts (Berg and McClaugherty 1987). The association between N release from the litter late in decay and the breakdown of the lignocellulose complex is not clearly understood, however, and deserves further study (Melillo and others 1989).

Despite the large biomass of woody debris in logs on the forest floor, the major supply of N for plants probably comes from the fine litter because its N concentration and content is so much higher than that of logs. Logs, however, should not be discounted in the N cycling story. Nitrogen fixation occurs in decomposing logs (Larsen and others 1980; Silvester and others 1982), particularly in decay classes II-IV, and it is important in the overall N balance of a site over the length of a rotation. Sollins and others (1987) noted that asymbiotic bacteria in fallen logs fix about 1 kg ha^{-1} yr^{-1} of N in the Oregon and Washington Cascades. This is a substantial amount relative to other system N inputs from precipitation and dry fall (2-3 kg N ha^{-1} yr^{-1}). Nitrogen in logs is available for fine roots and mycorrhizae to take up at much higher C:N ratios (> 300:1) than that in needles (20-30:1) and this, in combination with a favorable moisture regime, may explain why plants are able to grow so easily on CWD. Nitrogen release in Douglas-fir logs appears to coincide with the onset of lignin decay (Kermit Cromack, Department of Forest Science, Oregon State University, personal communicaton).

CONCLUSIONS

Large organic matter accumulations occur in western forests with higher accumulations in coastal than inland forests; higher productivity ecosystems have higher accumulations, particularly of CWD. Accumulations represent a balance among inputs, decomposition, and the effects of fire. Forest management has reduced the amount of coarse woody debris below that encountered in natural ecosystems, and this is of some concern with respect to the maintenance of long-term productivity in managed ecosystems.

Decomposition rates are much slower in inland than in coastal forests. Woody litter decomposes one to two orders of magnitude slower than needles. Despite its slow decomposition rate, however, woody litter provides a long-term source of soil organic matter and nitrogen, which could be important in maintaining site productivity. The decomposition process and its relation to forest productivity, however, is still incompletely understood.

Decomposition rates of litter are primarily determined by physical factors (especially temperature, moisture, and surface-to-volume ratios), substrate chemistry (especially lignin concentration), and biological factors (especially the presence of microbes and insects and plant species). Insects are particularly important in wood decomposition.

Clearcutting appears to increase the rate of fine litter decomposition, but may slow woody litter decomposition. Fine litter decomposition also appears more rapid in forest edges than in closed-canopy stands.

Nitrogen in fine surface litter is released relatively rapidly compared to release in CWD, where it is immobilized for many years. The rate of N release appears to be related to the rate of decomposition. Nitrogen does, however, become available to plants in large woody substrates at much higher C:N ratios (> 300:1) than in needle litter.

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Speakers answered questions from the audience after their presentations. Following are the questions and answers on this topic:

Q. (from Dave Gillman)—If the rates of decomposition are so slow, why isn't the organic accumulation on the soil surface thicker and why isn't there more woody debris in undisturbed areas?

A.—The amount of dead organic matter accumulating in a system is a balance among fine litter and coarse woody debris inputs, the rate of decomposition, and the effects of fire. In ecosystems with very slow decomposition because of moist, cold conditions, for example, at high elevations in the Washington Cascades, organic inputs to the soil are high and accumulatons on the soil suface are large, including large amounts of woody debris. Fire frequency is also low in these forests. In ecosystems that have a slow decomposition rate because they are hot and dry, organic inputs to the soil are low, resulting in low surface organic accumulations despite the low decomposition rate. Furthermore, frequent fire in these ecosystems prevents large organic accumulations.

CONNECTING FOREST PRODUCTIVITY TO BEHAVIOR OF SOIL-BORNE DISEASES

Geral I. McDonald

ABSTRACT

Armillaria-caused root disease in conifer forests of Western North America is discussed from an ecosystem perspective. Disease expression appears to be controlled by site-specific combinations of Armillaria *occurrence,* Armillaria *population reaction norms, host population reaction norms, and the environmentally conditioned interaction of these ecophysiological attributes. Adaptation and use of microclimate and ecological process models as a means of predicting these processes are presented. The role of models in ecosystem analysis is discussed and a method for calibrating the assimilation equations of an ecological process model to specific sites is presented. Simulations compare winter-active and winter-dormant chlorophyll conditions. These topics are then combined and integrated with a system of classification of habitat types into a temperature X moisture matrix designed to facilitate experimentation, validation, and construction of management options. Distribution of* Armillaria *species, clones within species, their ecological behavior, and some attributes of root disease epidemiology are related to the classification matrix.*

ONE VIEW OF FOREST HEALTH

Ecosystems composed of vigorous individuals genetically adapted to natural environments are healthy because stresses can be endured without strain. Endurance develops from preadapted mechanisms of stress avoidance, strain avoidance, or strain tolerance (see Levitt 1980 for definitions). Ecosystems may contain poorly adapted plants exhibiting higher levels of strain because their preadapted mechanisms are not synchronized with environmental stressors. Stress can develop from a variety of general sources: human changes such as air pollution (Smith 1984), or natural disturbances such as an extremely early fall frost, an abnormally cold spring, unusual mid-winter cold, or summer drought. Other possible stressors are exposure of plants to unnatural environments as a result of planting foreign seed sources, natural seeding of maladapted genotypes in heterogeneous environments, introduction of foreign pests, inappropriate management actions, and global change.

Effective management of both wild and domesticated ecosystems requires recognition of the strain caused by a specific stressor. Currently, our most common indicator of strain, unexpected mortality, is often attributed to the direct action of microbial or insect pests. However, the idea that these episodes in wild and supposedly "adapted" systems are triggered by stressed plants is gaining acceptance (Cates and others 1983; McDonald 1985; Shepard 1959; Waring and Pitman 1985). Diffuse mortality caused by *Armillaria* may result when scattered plants are stressed (Houston 1981; McDonald and others 1987a, 1987b; Wargo 1977, 1981). Accelerated mortality caused by *Armillaria* after stand entry for commercial thinning or other cutting operations (Filip 1977) is probably the result of strain increased by forest management activities (McDonald and others 1987b). It seems clear that subtle changes can have major impacts on system behavior.

The conifer/*Armillaria* pathosystem was chosen as the investigative model because of its widespread occurrence in a wide variety of interior West forest ecosystems (McDonald and others 1987a) and the detailed knowledge available about how various *Armillaria* species interact, at the level of the individual, with various host species to cause classic "stress disease." The role of stress in the woody plant/*Armillaria* pathosystem was recently given a thorough review (Wargo and Harrington 1991); but knowledge of specifically what stress creates strain, how to identify both, and how to manage a forest to prevent undue strain is still lacking. The objectives of this paper are to: (1) strengthen the ecophysiological maladaptation hypothesis of disease expression (McDonald and others 1987b) by a theoretical construction that ties the ideas of strain and genetically programmed physiological response together, (2) suggest an ecological framework suitable for testing hypotheses and organizing data about the function of root diseases in forest ecosystems, and (3) demonstrate use of the ecological framework in the conifer/*Armillaria* pathosystem as it functions in the montane forests of the Western United States.

FOREST HEALTH: A THEORETICAL CONNECTION

Cost-benefit analysis of adaptational strategies in plants (Solbrig 1979) is a key concept in formulating this hypothesis of how strain, in individuals or populations, conditions pest action. Plants as financial units (Horn 1979) have income (carbon assimilation) and expenses (respiration) such as maintenance, new growth (reproductive and vegetative), and defense (secondary compounds such as resin). Defense includes protection whenever needed against insect and pathogen pests plus physical stress, such as extremes of temperature, water availability, or light. Preparation for,

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or application of, defense is costly. Since plants are stationary, they must strike a genetic compromise between (1) ability to compete with neighbors (large and small), which is enhanced by shunting available income to growth, and (2) ability to survive pest attacks, wounds, and environmental extremes, which is enhanced by shunting income to maintenance of physiologic processes dedicated to specific or general defense mechanisms. These shunting activities can be flexible and thus responsive to environmental factors or fixed by adaptational forces. Evolution forges optimum compromises for given combinations of physical and biological environments.

Respiration can be partitioned into its maintenance and growth components (Amthor 1984; Marshall and Waring 1985). Plants need to satisfy the maintenance portion first (Amthor 1984). Whatever is left over can be used for growth. This is important because the partitioning pattern seems to have an inherited component (Amthor 1984) that can set limits on acclimation flexibility. If conifer populations adapt to some environmental components by increased levels of maintenance respiration, then plants so adapted would be at a competitive disadvantage when growing among plants adapted to an environment not requiring the higher maintenance rate because they would be less efficient. Ecotypic variation of root respiration in subalpine fir (ABLA) (see appendix for definition of species codes) and Englemann spruce (PIEN) (Sowell and Spomer 1986) demonstrates that such situations are possible in conifer populations.

REACTION NORM: THE JUNCTION OF GENE ECOLOGY AND ECOPHYSIOLOGY

A fundamental aspect of the ecophysiologic maladaptation view of host/pest interaction is the role of acclimative and adaptive tolerance in ecophysiologic and life history traits for both host and pest. The idea of acclimative tolerance was first proposed in 1909 by Worltereck (Stearns 1989), who defined the response of a genotype to an environmental stimulus as a reaction norm. A population can be viewed as a bundle of reaction norms (van Noordwijk 1989). Further, the width of the bundle indicates amount of heritable variation in the population and crossing reaction norms indicate genotype-environment interaction. This explanation of phenotypic plasticity was largely ignored until recently (Stearns 1989). Reaction norms can be thought of as the genotype's transformation of environmental variation into phenotypic variation (Stearns 1989).

Important to the present context is the idea that a reaction norm can "represent a nonadaptive or maladaptive response to unusual environments" (Stearns 1989). In other words, can normal evolutionary processes create and maintain maladapted populations? Maladaptation of a reaction norm (ecophysiological maladaptation) has been convincingly demonstrated in at least one case (Stearns and Sage 1980). A large population of mosquitofish (*Gambusia affinis*) living in brackish water appeared to hold a small population of the same species, living in nearby fresh water, in a state of physiological maladaptation. This situation seems analogous to my view of Rocky Mountain Douglas-fir (PSMEG) populations becoming adapted to the most prevalent environment (brackish-water mosquitofish) in a locality characterized by environmental heterogeneity and then producing genotypes that are sufficiently maladapted in many individual "islands" of dissimilar environments (fresh-water mosquitofish) that they become susceptible to *Armillaria* (McDonald in press).

Existence and strength of reaction norms in conifers and *Armillaria* can be assessed by measuring various genetic and phenotypic correlations among traits over various environmental gradients as has been done for plants (Antonovics 1976; Schlichting 1986) and insects (Via 1984). There is already some evidence that conifer populations will likely follow the same adaptational rules as other plants (Campbell 1979). Also, there is no reason to believe that *Armillaria* will behave differently.

If the pathosystem is working in this fashion, then one should expect variation in disease expression by combinations of management history and habitat types. Some important *Armillaria* attributes would be (1) clone size and age, (2) species and clone overlap, (3) species and clone ecophysiology, (4) species and clone pathogenicity, and (5) reaction norms of both life history and ecophysiological traits. Important host attributes are expected to be (1) photosynthesis profiles, (2) carbon allocation (maintenance vs. growth respiration), (3) water use profiles, (4) nutrient use profiles, and (5) reaction norms of life history and ecophysiological traits. These attributes are in turn influenced by specific factors of the site or ecosystem in which the pathosystem is operating. Some important ecosystem attributes are (1) annual air temperature, (2) annual precipitation, (3) annual precipitation pattern, (4) canopy and forest floor radiation profiles, (5) soil water availability profiles, (6) soil temperature profiles, (7) decomposition profiles, and (8) nutrient cycling profiles. The next question is—do these ideas have any experimental validity?

FOREST HEALTH/*ARMILLARIA*: AN EXPERIMENTAL CONNECTION

An hypothesis of physiologic interaction between *Armillaria* and hardwood hosts was developed by Wargo (1984). He showed experimentally that starch and sucrose concentrations drop while glucose, fructose, and free amino acids increase in roots of defoliated hardwoods (Wargo 1984). The same changes occur under drought stress (Parker 1979). Changes in lytic enzymes and ethanol may also occur, but seem to be of less importance (Wargo 1975). *Armillaria* growth in culture is enhanced by the presence of glucose (Garroway 1974) and the amino acid asparagine (Weinhold and Garroway 1966). Extracts from roots of defoliated trees (Wargo 1972) enhance growth. These findings are embodied in a hypothesis of physiologic interaction between *Armillaria* and hardwood hosts growing in northeastern North America (Wargo 1984). Successful colonization by a fungus in contact with healthy hosts requires the fungus to oxidize phenols (Wargo 1983). This ability is enhanced by increased glucose and free amino acids that occur with stress. Concurrently, reduced reserves diminish the host's ability to preserve its defenses.

An experimental connection between observed patterns of *Armillaria* behavior and host strain was reported by Redfern (1978). He conducted two stress experiments.

First, he grew *Armillaria*-inoculated *Larix kaempferi* (LAKA) under 16- or 8-hour photoperiods. The material grown under short days showed increased mortality. One explanation is short-day induced strain. Second, western hemlock (TSHE) and grand fir (ABGR) were inoculated with *Armillaria* and grown under full light or 70 percent shade. TSHE, a shade tolerant, showed equal mortality under both treatments, but for ABGR (less shade tolerant) mortality was increased from 44 to 76 percent.

Bourdeau and Laverick (1958) compared light-dependent curves of several conifers after growth under differing levels of shade. Eastern hemlock (TSCA) showed homeostasis, which is strain avoidance by acclimation, to the stress imposed by 70 percent shade (assumed little reduction of carbon production). If a conifer species cannot acclimate to low light or short days, the physiological consequences could be comparable to defoliation of a hardwood. We hypothesize that strain predisposed the LAKA and ABGR to *Armillaria*-caused injury in Redfern's experiment (1978), while TSHE avoided strain because acclimation prevented adjustment in carbon assimilation or allocation patterns.

Is the behavior of *Armillaria* in Western North America best explained by variation in the occurrence of strain in hosts or variation in the pathogenicity of the fungus? Pathogenic *A. ostoyae* exists east and west of the Cascade crest and on low- to high-productivity sites east of the crest. Reduced productivity in host populations is related to increases in disease for both interior to coast transects and from low- to high-productivity clines within interior forests.

The coastal region, where *Armillaria* is no problem on the same species as found inland, is characterized by massive annual carbon assimilation where 50 percent of the annual accumulation can occur between October and May (Bradbury and Malcolm 1978; Emmingham and Waring 1977; Waring and Franklin 1979). Winter assimilation capabilities east of the Pacific Northwest region are unknown. However, Scots pine (Oquist and Martin 1980) and other species (Bourdeau 1959; Jurik and others 1988; Perry and Baldwin 1966; Schulze and others 1967) have a period when chlorophyll is inactive. Frost hardiness shown by western white pine (PIMO) (Rehfeldt and others 1984) and PSMEG (Rehfeldt 1979) subjected to artificial freezing tests indicates winter-inactive chlorophyll.

Some conifer traits showing ecotypic variation (genetic response) are the light and temperature dependence of photosynthesis (Krueger and Ferrell 1965) and water relations (Ferrell and Woodard 1966; Jackson and Spomer 1979; Pharis and Ferrell 1966) and interactions of root respiration rates along soil temperature gradients (Sowell and Spomer 1986). Other woody plants have shown even more striking ecotypic variation (Berry and Bjorkman 1980; Pearcy and Harrison 1974). There is potential for creation of strain because of too little income to satisfy programmed expenses.

Regarding the direct connection of root rot behavior to soil conditions, there are published reports that soil parameters can influence the expression of *Armillaria*-caused disease. A connection was shown in an experimental situation with both inoculum survivability of *A. ostoyae* and infection of lodgepole pine (PICO) (Blenis and others 1989). A survey of occurrence of *Armillaria* infection in PSMEG showed a connection to low soil nitrogen and low pH (Shields and Hobbs 1979). The same study demonstrated a connection between ABGR and infection by *Armillaria* on low Ca and P and high K soils. Recently, Entry and others (1991) demonstrated links between low nitrogen and low light and increased infection of young PICO, PIPO, PSME, ABGR, and LAOC seedlings in a greenhouse experiment.

On the basis of points just reviewed: the known distribution patterns of potentially pathogenic *A. ostoyae*, the known ecotypic variations of key physiologic traits, Wargo's (1984) physiologic hypothesis, demonstrated direct connections to soil conditions (Blenis and others 1989; Shields and Hobbs 1979), and Redfern's (1978) and Entry and others' (1991) results, I hypothesize that ecosystem imbalances control expressions of the *Armillaria*/western-conifer pathosystem.

The ecophysiologic maladaptation hypothesis of disease expression implies the following assumptions: (1) *Armillaria* recycles woody plants under strain, (2) *Armillaria* species are genetically stable with clones thousands of years old and up to 800 meters in diameter (Shaw and Roth 1976), (3) hosts have adapted to various environments and to *Armillaria*'s forms, and (4) both parties exhibit inter- and intraspecies variation of acclimative and adaptive tolerances in ecophysiologic and life traits that interact or produce particular patterns of disease expression. Thus disease occurs only at suitable junctions of *Armillaria* occurrence, environment, host traits, and *Armillaria* traits. The challenge is to understand the interaction sufficiently to predict disease appearance and formulate effective management options.

ARMILLARIA IN WESTERN-MONTANE FORESTS

The genus is distributed throughout most but not all of the conifer forests of Western North America (McDonald and others 1987a). Millions of acres of western forests do not support *Armillaria* species because they are too dry. Some cold and wet forests are excluded from its range (McDonald and others 1987a). Inside its ecological boundaries, *Armillaria* shows a complex pattern of interaction with about two dozen conifer species. The pathogen varies from a well-behaved saprophyte through a mild pathogen to an aggressive pathogen. Individual host species vary from susceptible in virgin stands to resistant when growing in plantations placed on disturbed sites.

Evidence obtained so far indicates that several of the North American Biological Species (NABS) of *Armillaria* are distributed throughout this mix of forest types (McDonald unpublished data; McDonald and Martin 1988; Morrison and others 1985; Shaw and Loopstra 1988). At least one species, *Armillaria ostoyae* (NABS I) (McDonald 1991; Morrison and others 1985), is known to be pathogenic on conifers in these western forests. Additional species present in Western North America are *A. sinapina* (NABS V) (McDonald 1991; Morrison and others 1985; Shaw and Loopstra 1988); *A. gallica* (NABS VII) (McDonald 1991; Morrison and others 1985); NABS IX (species unknown) (Anderson and Ullrich 1979; Morrison and others 1985; Shaw and Loopstra 1988); NABS X (species unknown) (Anderson and Ullrich 1979; McDonald 1991; Morrison and others 1985); and *A. cepistipes* (NABS XI) (Morrison

and others 1985). Evidence from eastern Canada and the midwestern portion of the United States indicates that *A. gemina* (NABS II), *A. calvescens* (NABS III), *A. sinapina* (NABS V), and *A. mellea* (NABS VI) can also be pathogenic (Berube and Dessureault 1989; Proffer and others 1987).

No clear-cut relationship has been found relating pathogenic variants of the genus to specific host mortality (Raabe 1982). But some results indicate *Armillaria* may be virulent on specific classes of hosts. *A. mellea* is believed virulent on hardwoods in general and fruit trees in particular (Rishbeth 1982). *A. ostoyae* may be most aggressive to conifers but can colonize oaks (Davidson and Rishbeth 1988), and *A. gallica* may act as an ecosystem scavenger attacking either stressed hardwoods or conifers (Morrison and others 1985; Rishbeth 1982).

Armillaria behavior shows a clear distinction between coastal and interior regions of Western North America. The fungus is present throughout conifer forests of coastal British Columbia, Oregon, and Washington to the crest of the Cascade Mountains, but to date has caused no more than nuisance problems to young trees (Johnson 1976; Johnson and others 1972; Morrison 1981). East of the crest, *Armillaria* is a common problem over large areas (Filip and Goheen 1982; James and others 1984; Morrison 1981; Shaw and others 1976; Williams and Marsden 1982). *A. ostoyae* was identified as the responsible agent throughout interior British Columbia (Morrison and others 1985) at one site in the interior of Washington (Shaw 1984) and at several sites in northern Idaho and northeastern Oregon (McDonald 1991). Common occurrence of nonpathogenic *A. ostoyae* in coastal British Columbia forests was attributed to variation in pathogenicity (Morrison and others 1985).

An intensive survey of *Armillaria* occurrence and damage by habitat type and host species (McDonald and others 1987a, 1987b) in National Forests of the inland western United States presented several significant conclusions. First, trees on harsh sites showed higher incidence of pathogenic expression, on a plot basis, than those on milder sites. A word of explanation: "pathogenicity" as used in this paper relates to disease occurrence or absence on a 0.04-hectare plot. Criteria used to judge pathogenicity were (1) existence of mycelial fans in or under the bark on roots of living trees, (2) resin-soaked soil and fans in or under the bark on roots of dead trees, and (3) evidence of callus formation in association with old mycelial fans on dead trees. Within this region, site quality varies from almost as good as coast sites to very harsh. *Armillaria* was not found on the warm-dry and cold-dry sites. On the sites where it was found (species not yet identified), the most productive sites showed low incidence of pathogenicity, even after partial cutting. Sites of lesser productivity showed significantly higher probability of pathogenic *Armillaria* but only after stand entry. Finally, sites transitional to the cold-dry and warm-dry *Armillaria*-less sites showed high incidence of pathogenicity in pristine stands. With regard to species susceptibility, Douglas-fir (PSMEG), grand fir (ABGR), subalpine fir (ABLA), and Englemann spruce (PIEN) were most susceptible, but this ranking varied by site. Western hemlock (TSHE), western larch (LAOC), and western white pine (PIMO) were the least susceptible. Finally, *A. ostoyae* commonly occurred in a nonpathogenic mode on the most productive habitat types in northern Idaho (McDonald 1991). These results coupled with coast versus inland behavior indicate that *Armillaria* interaction with western conifers is likely mediated by host strain rather than variation in its innate ability to cause disease, as suggested by Morrison and others (1985).

CONNECTION TO WESTERN MONTANE ECOSYSTEMS

The western portion of North America is distinguished by complex landforms and associations of plant communities. Large variation in elevation, aspect, slope, latitude, and soil-forming materials has produced an elaborate mosaic of plant communities on the rugged landscape. Often widely differing communities reside side by side leading to local exchange of genes over wide environmental gradients. The situation has led to a complicated mixture of vegetation management problems not the least of which is native root rots. Because of its wide distribution and ecological amplitude, the genus *Armillaria* is one of the most important of the root-rotting organisms.

The interaction between causal agents and hosts that ultimately leads to expression of plant disease is always strongly governed by environment. This principle may be particularly true in the case of *Armillaria*-caused root rot as we have discussed. Thus, the logical place to begin charting a management path is a better understanding of the climate of the western forests. Representation of individual stands by way of annual temperature and precipitation averages and pattern of moisture accumulation is not satisfactory because of the wide range in moisture-holding capacity and thermal properties of soil-forming materials. For example, a ruling consideration in the Northern Rocky and Cascade Mountains is the distribution of volcanic ash. Fine ash distributed over the northern part of the Rocky Mountains increases the effective water-holding capacity to the point that annual precipitation of 85 cm, which produces a dry forest type in central Idaho (Steele and others 1981), produces a relatively moist forest type in northern Idaho (Cooper and others 1987). In regions outside the ash zone, such as western Oregon, an annual rainfall of 100 cm is required to produce a similar forest (Franklin and Dyrness 1973). An analogous situation applies to thermal conductivity of soils.

Given large variations in soil water-holding capacity, precipitation patterns, and thermal conductivity from place to place, one would expect that annual air temperature and precipitation would have only a modest chance of predicting effective soil moisture and temperature conditions. Add to this the great variation in radiation dynamics caused by canopy properties, slope, and aspect typical of western landforms and one can see that common indications of climate can be most inadequate and even misleading.

HABITAT TYPE CORRELATIONS

Forest ecologists have attempted to solve this dilemma by classifying forest sites according to associations of potential climax vegetation (Pfister and Arno 1980). This approach is believed to integrate effects of climate, topography, and to some extent soils in a single classification system. A great deal of effort has gone into descriptions

of plant associations all over the Western United States (Wellner 1989) and western Canada (Pojar and others 1987). Only the State of California lacks a classification system (Wellner 1989). Naming of habitat types makes use of plant species codes that consist of the first two letters of the genus and species names (appendix). We will use this convention to refer to all hosts, indicator species, and habitat types.

To assess the ecological behavior of *Armillaria* over all of Western North America, we need a regional correlation of habitat types (Bourgeron 1989). These correlations have not been completed. To temporarily satisfy this immediate need, we correlated the occurrence of some key indicator plants. A table based on four levels of annual average temperature and six levels of annual precipitation was constructed to serve as a correlative vehicle (fig. 1).

Twelve habitat types that have been connected to temperature and precipitation measurements (Cooper and others 1987; Franklin and Dryness 1973; Steele and others 1981; Youngblood and Mauk 1985) were then placed in their proper cells. The assignment of the 12 habitat types shows that several understory herbaceous and shrub indicators that have the requisite West-wide range seem to be more tied to moisture than temperature. They show strong tendencies to occur over large geographic and elevational ranges (fig. 1).

To assign habitat types to the proper temperature column, it was also necessary to establish some rules about the occurrence of conifer species as the climax overstory. Even though many conifer species occur throughout the western portion of North America, variation in their climax behavior is significant. Many of the conifer species of the West grow in the Northern Rocky Mountains of northern Idaho and northwestern Montana. This situation allows the establishment of climax guidelines. PIPO–ABGR–ABLA form a warm to cold sequence under dry conditions (Cooper and others 1987). Under slightly wetter conditions, PSMEG replaces PIPO. Under slightly drier conditions, ABCO replaces ABGR. Another increment of moisture allows THPL to dominate and start a warm to cold sequence of THPL, TSHE, ABGR, and ABLA (Cooper and others 1987). When the warm and wet climate of the west coasts of Oregon, Washington, and British Columbia is encountered, Sitka spruce (PISI) becomes a dominant seral species. Within the given ranges of annual rainfall as indicated by moisture-level-indicator plants, the conifer species appear as climax from warm to cold transects (fig. 1).

The occurrence of indicator plants and climax conifers in constancy tables from throughout the West (Brockway and others 1983; Cooper and others 1987; DeVelice and others 1986; Dyrness and Franklin 1974; Franklin and Dyrness 1973; Hanks and others 1983; Hemstrom and others 1987; Henderson and others 1989; Hoffman and Alexander 1983; Mauk and Henderson 1984; Moir and Ludwig 1979; Pfister and others 1977; Steele and others 1981, 1983; Topic and others 1988; Williams and Lillybridge 1983; Youngblood and Mauk 1985) was then used to assign other habitat types to the moisture levels of the matrix (fig. 2).

AVAILABLE-WATER INDEX IN CM	AIR TEMP IN DEGREES C: +8	5 TO 8	2 TO 5	-1 TO 2	AVAILABLE-WATER INDICATOR PLANTS
50 TO 75	NO TREES	PSMEG PIPO PIFL	ABCO PIEN	ABLA PIEN PIAL	JUCO PUTR RICE
DRY-SHRUB HABITAT TYPES					
75 TO 100	NO TREES	PSMEG PSMEM	ABGR ABCO	ABLA	DITR CHUM PAMY
DRY-FORB HABITAT TYPES					
100 TO 125	THPL	TSHE	ABGR ABCO TSME	ABLA TSME	CLUN COCA TITR
MOIST-FORB HABITAT TYPES					
125 TO 175	THPL	TSHE	ABGR TSHE	ABLA TSME	ASCA VIGL GYDR
WET-FORB HABITAT TYPES					
175 TO 225	THPL TSHE	TSHE THPL	ABGR TSME	ABLA TSME	ADPE ATFI BOVI
WET-FERN HABITAT TYPES					
225 +	THPL TSHE	TSHE THPL	ABGR TSHE	ABLA TSME	OPHO
WET-SHRUB HABITAT TYPES					

Figure 1—Moisture x temperature matrix used to define six classes of habitat types found in conifer forests of the Western United States.

AVAILABLE WATER IN CM	AIR TEMP IN DEGREES C: +8	5 TO 8	2 TO 5	-1 TO 2
50 TO 75	NO CONIFERS	PSMEG/JUCO PIPO/FEID PIPO/PUTR-FEID PIFL/FEID PIPO/PUTR	ABCO/FEAR ABCO/BERE ABCO/JUCO ABCO/ARPA PIEN/JUCO	ABLA/PIPU PIPU/CAFO PICO/SHCA PIEN/HYRE
75 TO 100	NO CONIFERS	PSMEG/PHMA PSMEM-LIDE/ARNE PSMEM/ARUV PSMEG/ACGL PSMEG/SYAL PSMEG/CARU	ABGR/LIBO ABCO/CEVE-CACH ABCO/HODI ABCO/CAGE ABGR/SPBE ABGR/XETE	ABLA/VAGL ABLA/LIBO ABLA/RIMO ABLA/VASC ABLA/VACA ABLA/XETE
100 TO 125	THPL/CLUN THPL-ABGR/ACTR	TSHE/CLUN TSHE/RHMA/BENE ABAM/BENE ABAM/GASH	ABGR/CLUN ABCO/SYAL-CLUN ABCO/CLUN TSME/VAAL	ABLA/MEFE ABAM-TSME/XETE ABLA/RHAL TSME/MEFE
125 TO 175	THPL/ASCA THPL/GYDR	TSHE/ASCA TSHE/ACCI-POMU	ABGR/ASCA ABAM/VAME-XETE	ABLA/CLUN TSME/CLUN
175 TO 225	THPL/ATFI TSHE/POMU THPL/ADPE	ABAM/TIUN TSHE-ABAM/LIBO	ABGR/SETR ABAM/ACTR	ABLA/STAM TSME/STAM ABLA/CACA
225 +	TSHE/POMU-OXOR TSHE-PISI/OPHO	THPL/OPHO ABAM/OPHO	CHNO/OPHO	ABLA/OPHO

Figure 2—Examples of defined habitat-types classified according to the matrix shown in figure 1.

AN INDEPENDENT CLASSIFICATION: CALIBRATED PROCESS MODELS

Verification and effective use of the habitat type classification system require independent assessments. Ideally, annual temperature and an available moisture index should be measured on specific plots of known habitat type. In the absence of such long-term records, I propose that a correlative microclimate model be used to reconstruct estimates of the needed long-term records. Also, if the climate model performs adequately, climate output can be used to drive ecosystem process models calibrated by site-specific physical and biological factors.

The major players are air and soil temperature, air and soil moisture, radiation, nutrient regimes, the living biomass, and the dead biomass. The initial need to understand and use a site-specific ecosystem approach is a daily representation of microclimate. This is available through the use of a mountain climate simulator—MTCLIM (Hungerford and others 1989). Next, one needs a way to integrate annual climate cycles into water balance regimes (Pojar and others 1987) and annual biomass production. A physiological process model (FOREST-BGC) suitable for conifer stands of the Northern Rockies has been developed (Running and Coughlan 1988) and tested (McLeod and Running 1988; Running and others 1987). Another model (MELTMOD) is available to simulate snow accumulation and melt (Leaf and Brink 1973). A model (LINKAGES) that simulates nitrogen dynamics in living and dead biomass has been deployed and tested for use in eastern forests (Pastor and others 1987). Needed but not available are (1) a model that links air temperature and moisture to decomposition and nitrogen fixation and to soil temperature models and (2) the linking of the available models into a functional whole.

These models will allow the interpretation and extension of knowledge about individual and stand indicators of strain such as sapwood basal area (Waring 1985), soluble amino nitrogen (Margolis and Waring 1986), growth efficiency (Waring 1985), starch content (McCullough and Wagner 1987), and leaf area index (Waring 1985). Another possible indicator of strain is the relationship between soil mineralizable nitrogen and stem growth rate (Powers 1980).

One way to verify such a system would be to collect sufficient physical and biological data from a number of individual plots to initialize various process and climatological models. The plots could then be classified according to their habitat type into the superclasses already defined (figs. 1, 2). We are attempting to connect the existing models to yield site-specific output. These models will be augmented by a soil temperature model (under development) and a nutrient cycling/decomposition model (under development). Performance of all models will be verified by collection of tree growth and weather data from field sites (McDonald and Andrews in press).

CALIBRATION OF A PROCESS MODEL TO A SPECIFIC SITE

For purposes of illustration, I will "calibrate" FOREST-BGC with photosynthesis parameters obtained from PSMEG seedlings grown under experimental conditions from seed obtained in northern Idaho (Koehn and McDonald these proceedings). Weather and site data are from a site in western Montana. The specific action of the chlorophyll moving from dormancy to activity and back to dormancy will be modeled after the data of Jurik and others (1988). The model will be driven by output from MTCLIM for a single year.

Before we start this exercise, let us take a quick look at verification of the combination of FOREST-BGC being driven by MTCLIM (McLeod and Running 1988). Six stands of PIPO growing in Montana were selected and their annual growth was measured. MTCLIM was initialized for the six stands and was run for the selected year to provide daily input of weather data to a version of FOREST-BGC. For the six stands, the correlation coefficient between predicted carbon assimilation and actual stand volume growth was 0.96 (McLeod and Running 1988).

MTCLIM is initialized by one or two regional weather records, average annual precipitation isohyet, east horizon, west horizon, elevation, slope, aspect, and leaf area index (Hungerford and others 1989). The model then supplies site-specific daily minimum and maximum air temperature, average relative humidity, radiation, and precipitation. FOREST-BGC is driven by the MTCLIM outputs and is initialized by soil, snow, latitude, and surface albedo physical parameters. It also requires biological parameters dealing with photosynthesis, respiration, water relations, leaf area index, and carbon allocation (Running and Coughlan 1988).

We began our attempts to calibrate the physiological process model by measuring parameters of photosynthesis. We have measured oxygen evolution and uptake with a system designed around the Hansatech leaf-disc oxygen electrode (Koehn and McDonald these proceedings; Walker and Osmond 1986). We measured five photosynthesis parameters as shown in a typical light response curve for a PSMEG seedling (fig. 6, Koehn and McDonald these proceedings). Three parameters are obtained from the linear portion of the curve (fig. 3, Koehn and McDonald these proceedings) and two more are obtained from an analysis of the kinetics of oxygen evolution response to light after oxygen used in photorespiration is added (fig. 5, Koehn and McDonald these proceedings). Three of the four parameters are used in the equation in FOREST-BGC (Running and Coughlan 1988) that calculates assimilation (fig. 7, Koehn and McDonald these proceedings).

To calibrate the assimilation equation used in FOREST-BGC, the following modifications and assumptions were made. The equation used in the model is:

$$\mu MOL\ CO_2 M^{-2} S^{-1} = \frac{ICCxMCCxMMCxLMCS}{MCC+MMCxLMCS}$$

where

ICC = internal CO_2 concentration = 59,259 μbars = 5 percent CO_2 in Hansatech cells
MCC = maximum stomate conductance = 0.008 MS^{-2} (Lohammer and others 1980)
MMC = maximum mesophyll conductance = PMAX/ ICC
LMCS = light-dependent mesophyll conductance scaler.

The term *LMCS* is composed of the following equation used by Running and Coughlan (1988):

$$LMCS = \frac{Q - Q_o}{Q + Q_{0.5}}$$

where

Q = current radiation level
Q_o = light compensation point
$Q_{0.5}$ = radiation level where mesophyll conductance is 0.5 of maximum.

Some assumptions were necessary to enable the use of measured parameters. First, *MMC* was related to *PMAX* (maximum rate of photosynthesis). If we assume that *PMAX* is related to mesophyll conductance according to the equation of Landsberg (1986):

$$A_{max} = k_c(C_i - \Gamma$$

where

A_{max} = light saturated assimilation rate
k_c = carboxylation efficiency
C_i = internal leaf CO_2 concentration
Γ = rate of CO_2 efflux in the light at zero C_i in C_3 plants.

According to Landsberg (1986) k_c = mesophyll conductance and Γ can be ignored. Thus, given that A_{max} can be measured and C_i was computed by FOREST-BGC or could be calculated, *MMC* was estimated by

$$MMC = \frac{PMAX}{ICC}$$

The term $Q_{0.5}$ is equivalent to the Michalelis-Menten (M-M) constant. Some argue against the analogy between photosynthesis as a biochemical rate of substrate conversion and photon flux density as a substrate (Causton and Dale 1990) and enzyme kinetics. But the M-M model seems to adequately fit the O_2 evolution data (Koehn and McDonald these proceedings) and the computed parameters calibrate existing models of photosynthesis. Thus, the M-M constant was substituted for $Q_{0.5}$ in the assimilation equation. The last parameter estimated was light compensation point. A method of measurement was given elsewhere (figs. 3, 4, Koehn and McDonald these proceedings). For illustration, assume that PSMEG 144 represents trees having winter-active chlorophyll and PSMEG 123 represents trees having winter-dormant chlorophyll. Parameters were translated from O_2 to CO_2 basis by assuming uptake is 0.75 of evolution.

The actual behavior of PSMEG with regard to winter chlorophyll behavior is unknown. For illustration we assumed that this species would behave similarly to eastern white pine (PIST) in Michigan (Jurik and others 1988). I ran MTCLIM for a site located at 4,500 feet near Missoula, MT, for the year 1984 and computed a plot of accumulated degree days above 1 °C, and applied the PIST chlorophyll triggers (fig. 3). Our preliminary data show that the rate constant of photosynthesis, based on the M-M model, might be reduced in winter-dormant chlorophyll (Koehn and McDonald these proceedings). I also assumed that maximum mesophyll conductance would be reduced independent of its effect on *PMAX*. The values and changes from dormant to active are shown in figure 4.

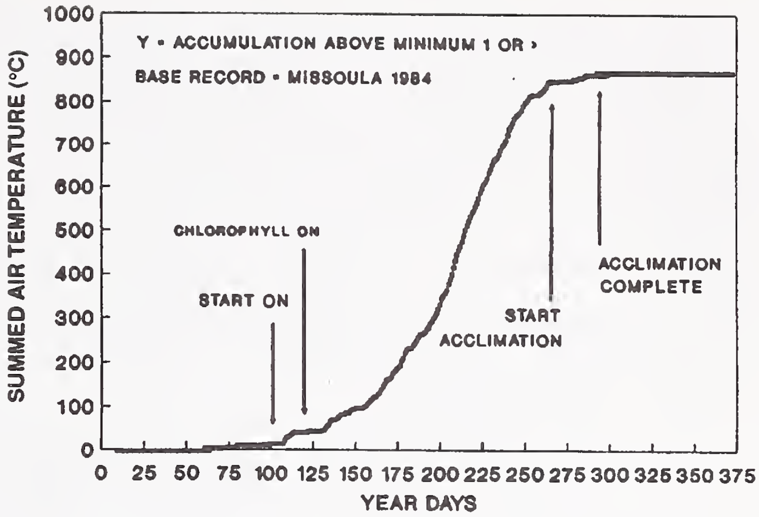

Figure 3—Assumed chlorophyll activity of PSMEG at a site near Missoula, MT, as a function of accumulated daily minimum temperatures above 1 °C as simulated by MTCLIM. Triggers on and off patterned after PIST (Jurik and others 1988).

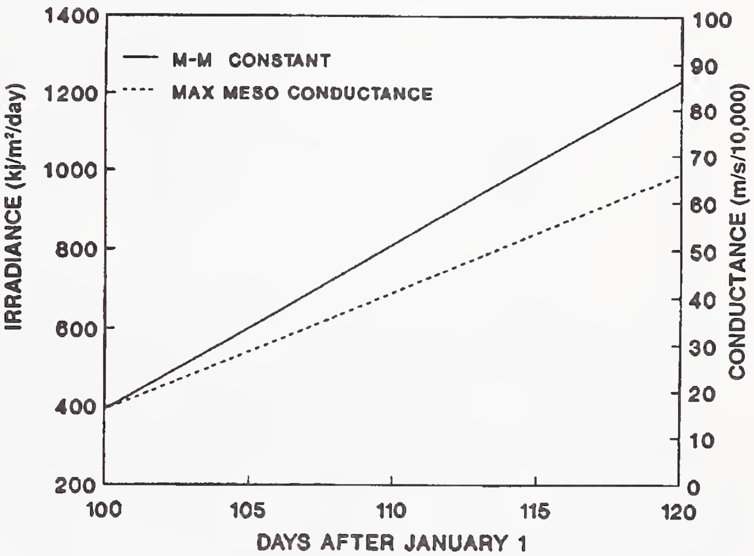

Figure 4—Assumed springtime chlorophyll recovery function of PSMEG for rate constant of oxygen evolution over light and maximum mesophyll conductance.

An important site-specific ecological component is the annual available soil moisture profile. I assumed the initial values shown in figure 5 and ran FOREST-BGC for north and west aspects at the Missoula, MT, site (fig. 5). The difference in water use profile is due to the difference in snowpack. All other parameters were equal.

The complete model was run with physical initial parameters set for the west slope and the tree parameters set for winter active and winter dormant chlorophyll (fig. 6). According to this simulation, most of the assimilation at the site was completed by June 15. It also shows a large difference in annual carbon assimilation accumulated by the winter active population through March, April, and May. The possible importance of winter-dormant chlorophyll is seen in the annual assimilation simulations for the two slopes and three chlorophyll conditions (fig. 7).

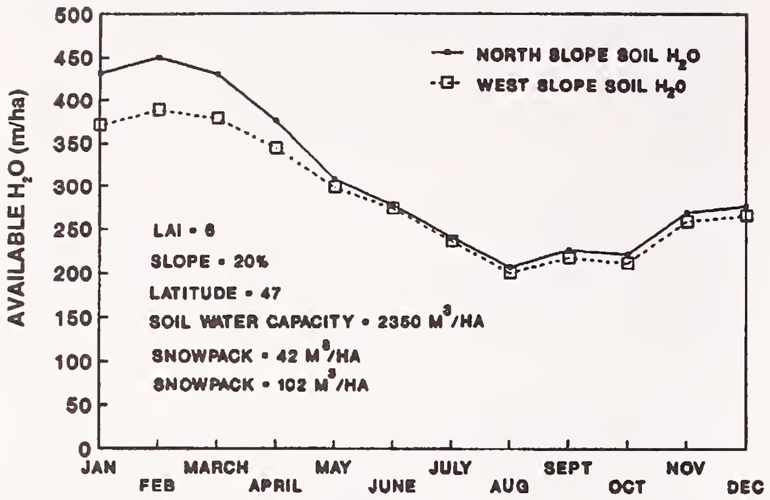

Figure 5—FOREST-BGC simulated water use curves for north and west aspects for a site near Missoula, MT. Soil water capacity was estimated from soil maps and snowpack quantities were assumed.

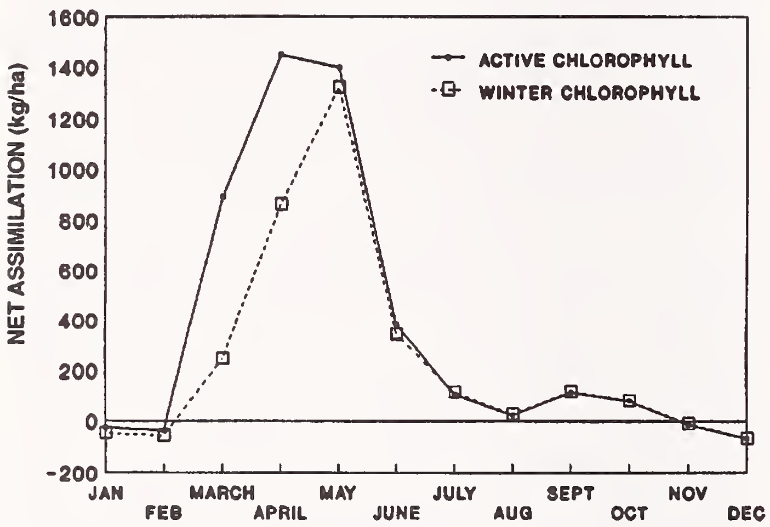

Figure 6—FOREST-BGC simulated monthly assimilation as calibrated by laboratory measurement of photosyntheses parameters under standard conditions (Koehn and McDonald these proceedings). Simulation was based on north slope weather regenerated by MTCLIM from same base station and initial parameters as figure 5.

ARMILLARIA BEHAVIOR BY HABITAT TYPE

The classification of habitat types discussed earlier might be a useful tool to analyze the ecological behavior of *Armillaria* species in the forests of Western North America (fig. 1). We know considerable about the distribution of *Armillaria* in general terms over much of Western North America and a substantial amount about distribution and behavior by habitat type in the Northern Rocky Mountains (McDonald 1991). With this knowledge we can correlate the occurrence of NABS and other specific ecological behavior patterns of the pathogen to ecophysiologic and adaptational attributes of host species in disrupted as well as virgin stands.

Properties that could be important in assessing the potential for disease expression are frequency of stump colonization by host and *Armillaria* species, frequency and pattern of kills, impact of inoculum buildup in stumps on either natural or planted replacement stands, and pattern of damage (scattered or clumped) in virgin and disturbed stands.

Several ecological attributes of *Armillaria* species and their clones could influence these epidemiological considerations and be associated with habitat type. These are: (1) extent of species and clone overlap, (2) extent and density of species and clone occupancy of stands before and after disruption, (3) average geographic extent of species and individual clones, (4) frequency of epiphytic rhizomorphs on vigorous trees, (5) relative frequency of haploids and diploids, (6) relative pathogenicity of species and clones, and (7) susceptibility of conifer hosts. Another *Armillaria* attribute possibly associated with habitat type is relative importance of root-to-root contact and rhizomorphs in disease spread. Understanding these attributes and patterns leads to conclusions about current and future ecological roles of the various *Armillaria* species and provides the foundation for charting a reliable management path.

Risks connected with planting exotic species, offsite seed sources, or improved seed could be predicted by use of indicator plants and the ecological process models. In addition the effects of massive site degradation by very hot fire, severe logging practice, and severe site preparation could be predicted. Effects of slow and long-term insults such as removal of nutrient capital, addition of site-degrading amendments, and climate change may be predictable as well. The system will be illustrated by incorporating a number of plots where we have determined habitat types and *Armillaria* behavior into the classification scheme (fig. 1). *Armillaria* species were identified by a diploid x diploid cultural mycelial challenge (McDonald and others 1991).

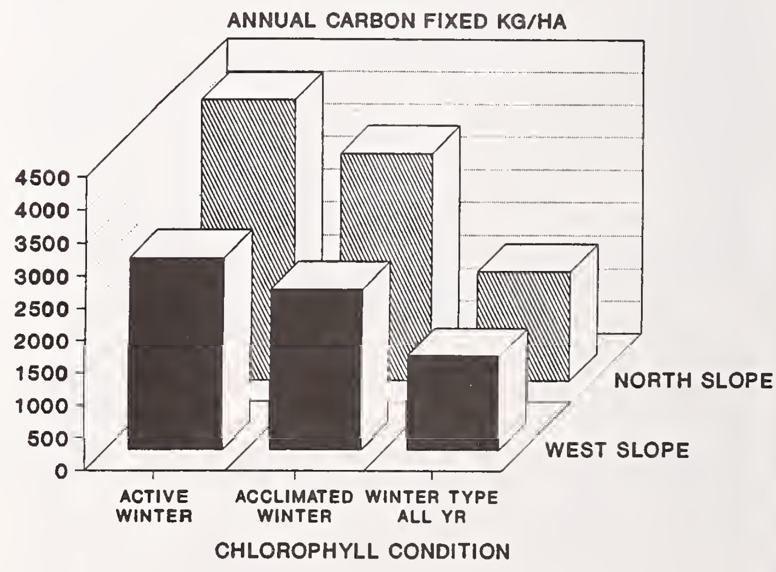

Figure 7—FOREST-BGC simulated annual net assimilation for six combinations of chlorophyll activity and aspect as expected from calibration by photosyntheses parameters measured under standard conditions for PSMEG seedlings.

Dry-Shrub Habitat-Type Group

Some of the shrub species that indicate dry and hot distribution limits of the genus *Armillaria*, by their use in habitat type names even though their actual distribution may exceed that of the habitat type, are JUCO (common juniper), PUTR (bitterbrush), and RICE (squaw current). The group of habitat types that include these species (see fig. 8) occupy the driest forest sites on flat and southerly exposures from low to high elevations throughout the entire Western United States. The exact amount of acreage has not been determined, but it encompasses most of the PIPO climax, a relatively small amount of the PSMEG climax, about 25 to 50 percent of the ABCO climax, and less than 25 percent of the ABLA climax. Only three plots in this group (fig. 8) have been sampled for the fungus, and none produced evidence of it. Also, reports of the fungus in these dry western forests by others are rare.

Since this group represents millions of acres of forest land, it should be sampled more thoroughly using the techniques outlined in this paper to determine risk to the fungus. Also, the occurrence of any *Armillaria* in these habitat types should be studied to better define reaction norms and potential for adaptation to dry conditions. The opportunity exists to label millions of acres as having very low potential for *Armillaria* damage.

DRY SHRUB GROUP OF HABITAT TYPES					
CLIMAX		HARDWOOD	PSMG-PIPO	ABCO-PIEN	ABLA-PIEN
# PLOTS		0	3	0	0
BEHAVIOR[1]		O P S	O P S	O P S	O P S
NABS	I	NO DATA	[2]0	NO DATA	NO DATA
	V		0		
	VII		0		
	IX		0		
	X		0		
	UNK		0		

DRY FORB GROUP OF HABITAT TYPES					
CLIMAX		HARDWOOD	PSMG-PSMM	ABGR-ABCO	ABLA
# PLOTS		0	9	8	14
BEHAVIOR		O P S	O P S	O P S	O P S
NABS	I	NO DATA	3 3 0	6 5 1	0
	V		0	0	0
	VII		0	0	0
	IX		0	0	0
	X		0	1 0 1	0
	UNK		0	5 1 4	0

[1] O = OCCURRENCE P = PATHOGENIC S = SAPROPHYTIC
[2] NUMBER OF CLONES

Figure 8—Behavior and distribution of *Armillaria* intersterility groups and clones according to habitat types classified into dry-shrub and dry-forb groups.

Dry-Forb Habitat-Type Group

Perennial forbs found growing on somewhat more moist situations that indicate the likely presence of *Armillaria* in some PIPO, PSMEG, ABGR-ABCO, and ABLA climaxes are ADBI (trail-plant), DITR (fairybells), GABO (northern bedstraw), PYSE (sidebells pyrola), SMST (starry solomon's seal), SMRA (feather solomonplume), THOC (western meadowrue), GOOB (western rattlesnake plantain), and MIST (side-flowered mitella). Some moist site shrubs can also signify this group. These are PAMY (Oregon boxwood), CEVE (snowbrush), CHUM (Prince's pine), ROGY (baldhip rose), and RUPA (western thimbleberry).

Any combination of these shrubs and forbs indicates the probable existence of *Armillaria* in the habitat type. The wet end of this group appears to have nearly 100 percent constancy of the fungus while the dry end may have less than 10 percent constancy (McDonald 1991). In any case many more plots need to be established in this group. Evidence to date shows that the fungus is an important consideration in these habitat types (fig. 8). Nine 0.1-acre plots have been sampled on the PSMEG series, and three yielded only single pathogenic clones of *A. ostoyae*. My general belief is that about 50 percent of the 0.1-acre plots in this class will support the fungus and, if it has not already expressed itself in virgin stands, it probably will with any stand disturbance. Stands in this category should be considered at high risk for PSMEG. Natural PIPO seems to be very disease tolerant, and LAOC may also be a disease-tolerant option on the wet end of this class.

The ABGR-ABCO potential climax series of this group shows a different pattern of response (fig. 8). Six of eight plots had NABS I, and in five cases it was pathogenic. On the sixth, it was saprophytic. In the first five cases a single clone was found. On the sixth plot, one pathogenic and one saprophytic clone were found. For two cases on the east slope of the Cascade mountains in Washington and Oregon and one case in northern New Mexico, NABS I was pathogenic on natural PIPO. In one case in western Montana, a single clone was pathogenic on natural PSMEG, PIPO, and ABGR. These four plots had a history of partial cutting.

Two of the plots were undisturbed, located in northern Idaho, and supported NABS I pathogenic on PSMEG. One of these plots also supported a saprophytic clone of NABS I. One plot had a single saprophytic clone of NABS X. Another plot had one pathogenic clone of an unknown NABS (not yet positively identified) on PSMEG. Finally, I have inspected 14 plots in ABLA series of the dry forb group. None have supported any *Armillaria* species.

Moist-Forb Habitat-Type Group

Perennial forbs indicative of sufficient soil moisture to support THPL and TSHE make their appearance in this group. The most widespread and useful of these are CLUN (queencup beadlily), TIRT (coolwort foamflower), and COCA (bunchberry dogwood). These forbs indicate the best ABCO, ABGR, and ABLA climax stands and the most moisture-limited of the THPL and TSHE climax stands. Again, the idea is to key on the presence of the indicator species in its role as habitat type indicator, not on its absolute distribution. These species frequently occur on more moist habitat types.

Armillaria ostoyae is nearly always present and occasionally is pathogenic in virgin stands on natural PSMEG, ABGR, ABLA, and PIEN but not to natural PIPO, PIMO, TSHE, THPL, or LAOC. Pathogenicity is often expressed to natural and planted PSMEG, planted PIMO, and planted LAOC in disrupted stands (McDonald 1991). Natural TSHE and THPL are seldom susceptible when growing in plantations of other species. In general, NABS X is nearly always present (12 of 13 plots) as a saprophyte, and NABS I is sometimes present as a pathogenic clone in the TSHE series (fig. 9). NABS I is frequently pathogenic in the other series (fig. 9).

Wet-Forb Habitat-Type Group

Perennial forbs that indicate relatively wet or high water-holding capacity soils and generally well-drained and aerated situations signify the next increment of available moisture. These high site indicators are present only in the forests of western Oregon, western Washington, western British Columbia, the southern coast of Alaska, some areas of eastern Washington, northern Idaho, interior British Columbia, western Montana, and the northern coast of California. The most widespread of these indicator plants are ASCA (wild ginger) and VIGL (pioneer violet). A more restricted indicator of this group is GYDR (oak fern).

Probability of encountering the principal pathogenic species of the Western United States, *A. ostoyae*, appears to be dropping within this class. Further, if present, it is a saprophyte. Consequently, PSMEG, ABGR, ABLA, and PIEN are at small risk on both virgin and disrupted stands. Saprophytic NABS X is still present, and an unknown pathogenic clone was observed (fig. 9).

Wet-Fern Habitat-Type Group

The next class of available moisture is based on the presence of three species of ferns: ATFI (ladyfern), ADPE (maidenhair fern), and BOVI (moonwart). Stands containing these plants exist in western Oregon, Washington, coastal British Columbia and Alaska, eastern Washington, northern Idaho, and northwestern Montana.

The warmest portion of this group is not currently represented by our sampling (fig. 10). The next series class is represented by three 0.1-acre plots in virgin stands on the Clearwater and Nez Perce National Forests of northern Idaho. Similar plant communities are found in western Oregon, Washington, and British Columbia. Two of these plots were classed as THPL/ADPE. On one plot, a single unknown clone was epiphytic on ABGR, PSMEG, PIMO, ACGL, and AMAL (McDonald 1991). On another, a single clone of *A. ostoyae* was epiphytic and saprophytic on ABGR, PSME, THPL, and BEPA. A clone of NABS X was epiphytic on hardwood brush. Also inspected was a plot in an old-growth THPL/ATFI stand. This plot supported a single epiphytic clone of NABS X.

The habitat types falling in the colder range of the wet-fern group are mostly limited to the forests west of the

MOIST FORB GROUP OF HABITAT TYPES

CLIMAX		THPL			TSHE			ABGR-ABCO			ABLA-TSME		
# PLOTS		2			6			2			3		
BEHAVIOR[1]		O	P	S	O	P	S	O	P	S	O	P	S
N	I	[2]2	2	0	7	2	5	1	1	0	2	2	0
A	V	1	0	1	0			0			0		
B	VII	0			0			0			0		
S	IX	0			0			0			0		
	X	2	0	2	5	0	5	3	0	3	1	0	1
	UNK	0			1	0	1	0			0		

WET FORB GROUP OF HABITAT TYPES

CLIMAX		THPL-TSHE			TSHE-THPL			ABGR-TSHE			ABLA-TSME		
# PLOTS		4			5			2			0		
BEHAVIOR		O	P	S	O	P	S	O	P	S	O	P	S
N	I	0			2	0	2	0			NO DATA		
A	V	0			0			0					
B	VII	1	0	1	0			0					
S	IX	0			0			0					
	X	5	0	5	5	0	5	3	0	3			
	UNK	2	1	1	0			2	0	2			

[1] O = OCCURRENCE P = PATHOGENIC S = SAPROPHYTIC
[2] NUMBER OF CLONES

Figure 9—Behavior and distribution of *Armillaria* intersterility groups and clones according to habitat types classified into moist-forb and wet-forb groups.

WET FERN GROUP OF HABITAT TYPES

CLIMAX		THPL-TSHE			THPL-TSHE			ABGR-TSME			ABLA-TSME		
# PLOTS		0			3			0			0		
BEHAVIOR[1]		O	P	S	O	P	S	O	P	S	O	P	S
N	I	NO DATA			[2]1	0	1	NO DATA			NO DATA		
A	V				0								
B	VII				0								
S	IX				0								
	X				2	0	2						
	UNK				1	0	1						

WET SHRUB GROUP OF HABITAT TYPES

CLIMAX		THPL-TSHE			TSHE-THPL			ABGR-TSHE			ABLA-TSME		
# PLOTS		0			0			0			0		
BEHAVIOR		O	P	S	O	P	S	O	P	S	O	P	S
N	I	NO DATA			NO DATA			NO DATA			NO DATA		
A	V												
B	VII												
S	IX												
	X												
	UNK												

[1] O = OCCURRENCE P = PATHOGENIC S = SAPROPHYTIC
[2] NUMBER OF CLONES

Figure 10—Behavior and distribution of *Armillaria* intersterility groups and clones according to habitat types classified into wet-fern and wet-shrub groups.

Cascade crest. Consequently, they have not been sampled for *Armillaria*, and no information relative to this topic is available.

We have limited specific information on which to base conclusions for this group, but some general perceptions may help us out. Since even the inland representatives are coastlike ecosystems, they should correspond to the situation found in the forests west of the Cascade crest. Random surveys of *Armillaria* behavior have not been conducted, but forest pathologists working on the west side indicate that damage caused by *Armillaria* is very limited (Goheen 1990). A survey based on *Armillaria* sporophores was conducted for all of British Columbia (Morrison and others 1985). This survey revealed that *A. ostoyae* was widely distributed in both coastal and interior forests. Since these forests are very similar to those in Washington, Oregon, and Idaho, we can conclude that this pathogenic species is also widely distributed in the West Coast forests of the United States.

The question is—what is the reason for the absence of pathogenicity in the two locations? The gradual decrease in aggressive pathogenic behavior of *A. ostoyae* as one moves to more productive and presumably less stressful ecosystems argues for changes in host stress as the fundamental factor. The case histories give ample testimony to the widespread existence of potentially pathogenic clones residing in a epiphytic or saprophytic state. The opposing view is that this difference is caused by pathogenic variation in the fungus (Morrison and others 1985). At any rate, the coastlike behavior pattern of *A. ostoyae* observed in the coastlike ecosystems of the interior that are surrounded by seas of pathogenic *A. ostoyae* seems to argue for a strain-mediated interaction.

Wet-Shrub Habitat-Type Group

The wettest group of habitat types is signified by the presence of OPHO (devil's club). This shrub seems to occur across the entire temperature range just as do most of the other indicator plants. No specific plots in the group have been surveyed for the explicit connection of habitat type and *Armillaria* behavior. There are some general studies that can be interpreted in terms of the structure outlined above.

First, the unnamed NABS IX was found only in the warmest and wettest part of British Columbia (Morrison and others 1985) and along the coast in Alaska (Shaw and Loopstra 1988). It was not found in the interior upland collections of the Rocky Mountains (McDonald 1991). Samples have been found in northern Idaho (Anderson and Ullrich 1979), but the most likely habitat type (THPL/OPHO) has not been searched.

The most abundant species found in coastal Alaska is *A. sinapina* (Shaw and Loopstra 1988). It is relatively rare in the interior of the Pacific Northwest (McDonald 1991). Finally, one must consider the possibility that our most pathogenic species, *A. ostoyae*, does not occur in the warmest and wettest of western forests. It was not found in coastal Alaska (Shaw and Loopstra 1988), and the evidence from the interior indicates a reduction in the number of clones per plot as the forests become wetter. The central question here is—what is the constancy of *A. ostoyae* in the habitat types of the wet-shrub group and where is its ecological boundary? Perhaps very warm, very dry, and very wet habitat types can be placed outside the sphere of *Armillaria* problems.

CONCLUSIONS

Knowledge about traits that limit the geographic range of *Armillaria* is of even greater interest than explicit knowledge about the geographic range itself. What are its reaction norms over a variety of environmental gradients, and how do these impact its pathogenic capabilities? What are the shapes of various reaction norms of the host species, and how do they interact with reaction norms of the pathogen to produce disease? Our increasing understanding of pathogen distribution and ecology coupled with increased understanding of the adaptational architecture of hosts and pathogen will lead to creation of more-focused experiments. Experiments conducted from the viewpoint of the ecophysiologic maladaptation hypothesis should enable the design of a West-wide *Armillaria* management plan flexible enough to predict system responses to changing conditions and simultaneously show the way to effective management of the modified system. This approach might also serve as a model for creating management plans for additional root rots and other major forest pests.

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APPENDIX: SPECIES CODES, SCIENTIFIC NAMES, AND COMMON NAMES OF PLANTS MENTIONED IN THIS PAPER

Tree Species

ABLA, *Abies lasiocarpa*, subalpine fir
PIEN, *Picea englemannii*, Englemann spruce
PSMEG, *Pseudostuga menziesii* var. *glauca*, Rocky Mountain Douglas-fir
PSMEM, *Pseudostuga menziesii* var. *menziesii*, Douglas-fir
THPL, *Thuja plicata*, western redcedar
TSHE, *Tsuga heterophylla*, western hemlock
TSCA, *Tsuga canadensis*, eastern hemlock
TSME, *Tsuga mertensiana*, mountain hemlock
LAKA, *Larix kaempferi*, Japanese larch
LAOC, *Larix occidentalis*, western larch
ABCO, *Abies concolor*, white fir
ABGR, *Abies grandis*, grand fir
ABAM, *Abies amabilis*, Pacific silver fir
CHNO, *Chamaecyparis nootkatensis*, Alaska-cedar
PIMO, *Pinus monticola*, western white pine
PICO, *Pinus contorta*, lodgepole pine
PIPO, *Pinus ponderosa*, ponderosa pine
PIFL, *Pinus flexilis*, limber pine
PIPU, *Picea pungens*, blue spruce
PISI, *Picea sitchensis*, Sitka spruce
LIBE, *Libocedrus decurrens*, incense-cedar

Shrub Species

JUCO, *Juniperus communis*, common juniper
RICE, *Ribes cereum*, squaw current
PUTR, *Purshia tridentata*, bitterbrush
PAMY, *Pachystima myrsinites*, Oregon boxwood
CEVE, *Ceanothus velutinus*, snowbrush
CHUM, *Chimaphila umbellata*, prince's pine
ROGY, *Rosa gymnocarpa*, baldhip rose
RUPA, *Rubus parviflorus*, western thimbleberry
LIBO, *Linnaea borealis*, twinflower
OPHO, *Oplopanax horridum*, devil's club
BERE, *Berberis repens*, Oregon grape
ARPA, *Arctostaphylos patula*, greenleaf manzanita
SHCA, *Shepherdia canadensis*, russet buffalo-berry
ARNE, *Arctostaphylos nevadensis*, pinemat manzanita
ARUV, *Arctostaphylos uva-ursi*, bearberry
ACGL, *Acer glabrum*, Rocky Mountain maple
SYAL, *Symphoricarpos albus*, snowberry

(con.)

CACH, *Castanopsis chrysophylla*, golden chinkapin
HODI, *Holodiscus discolor*, ocean spray
SPBE, *Spiraea betulifolia*, white spirea
VACA, *Vaccinium caespitosum*, dwarf huckleberry
VAME, *Vaccinium membranaceum*, big huckleberry
VASC, *Vaccinium scoparium*, grouse huckleberry
VAAL, *Vaccinium alaskaense*, Alaska huckleberry
VAGL, *Vaccinium globulare*, globe huckleberry
MEFE, *Menziesia ferruginea*, fool's huckleberry
RIMO, *Ribes montigenum*, mountain gooseberry
RHAL, *Rhododendron albiflorum*, Cascade azalea
RHMA, *Rhododendron macrophyllum*, rhododendron
BENE, *Berberis nervosa*, dwarf Oregon grape
GASH, *Gaultheria shallon*, salal

Forb Species

ADBI, *Adenocaulon bicolor*, trail-plant
DITR, *Disporum trachycarpum*, fairybells
GABO, *Galium boreale*, northern bedstraw
PYSE, *Pyrola secunda*, sidebells pyrola
SMST, *Smilacina stellata*, starry solomonplume
SMRA, *Smilacina racemosa*, feather solomonplume
THOC, *Thalictrum occidentale*, western meadowrue
GOOB, *Goodyera oblongifolia*, western rattlesnake-plantain
MIST, *Mitella stauropetala*, side-flowered mitella
CLUN, *Clintonia uniflora*, queencup beadlily
TITR, *Tiarella trifoliata*, coolwort foamflower
COCA, *Cornus canadensis*, bunchberry dogwood
ASCA, *Asarum caudatum*, wild ginger
VIGL, *Viola glabella*, pioneer violet
XETE, *Xerophyllum tenax*, beargrass
TIUN, *Tiarella trifoliata* var. *unifoliata*, coolwort foamflower
ACTR, *Achlys triphylla*, vanilla leaf
STAM, *Streptopus amplexifolius*, twisted-stalk
SETR, *Senecia triangularis*, arrowleaf groundsel

Fern Species

GYDY, *Gymnocarpium dryopteris*, oak-fern
ATFI, *Athyrium filix-femina*, ladyfern
ADPE, *Adiantum pedatum*, maidenhair fern
BOVI, *Botrychium virginianum*, moonwort
POMU, *Polystichum munitum*, western swordfern

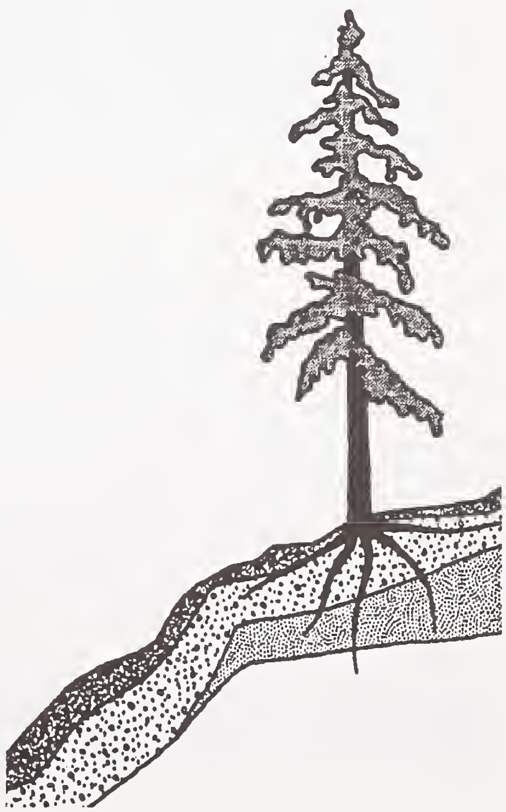
Grass Species

CACA, *Calamagrostis canadensis*, bluejoint reedgrass
CAFO, *Carex foenea*, spruce sedge
FEAR, *Festuca arizonica*, Arizona fescue
FEID, *Festuca idahoensis*, Idaho fescue
CAGE, *Carex geyeri*, elk sedge
CARU, *Calamagrostis rubescens*, pinegrass

Moss Species

HYRE, *Hypnum revolutum*, hypnum

Soil Alterations, Damage, and Restoration



EROSION AND SITE PRODUCTIVITY IN WESTERN-MONTANE FOREST ECOSYSTEMS

Walter F. Megahan

ABSTRACT

Soil loss from erosion affects site productivity by reducing the nutrient pool and water-holding capacity of the soil and by direct damage to vegetation. The effects of erosion depend on the type of erosion processes because of differences in the depth and areal extent of soil loss, the downslope rate of soil movement, and the probability of redeposition of eroded material. The most severe and long-lived site productivity losses occur from debris avalanches and gullying. Forest management practices can increase erosion rates, but wildfires have the greatest potential to accelerate erosion. Erosion increases following fire are directly proportional to fire intensity. Debris landslides and gullying cause serious and long-term reductions in site productivity, but the areas affected are small. Surface erosion occurs over much larger areas and does tend to reduce site productivity, but the magnitude of the reduction is poorly defined because of the compounding effects of compaction on logged areas and water repellency on burned areas. Methods to better assess the erosional effects of forest management on site productivity require a combination of controlled bioassay studies and growth-simulation models.

INTRODUCTION

Erosion is a geomorphic process that is a natural component of any forest ecosystem. However, erosion rates can be accelerated by both natural and human disturbances. Wildfire is the most common cause of accelerated erosion in the "natural" forest. Forest management activities, especially timber harvest and road construction, have been shown to increase erosion rates on forest lands. Megahan (1981) summarized the results of 30 studies documenting the effects of fire and forest management practices on erosion rates in the western-montane region.

To date, the largest concern with accelerated erosion in relation to forest management has been directed at the resulting downstream sedimentation and accompanying damage to beneficial uses of water. However, concern about the onsite impacts of erosion on forest lands is increasing. The Pacific Northwest Region of the Forest Service, U.S. Department of Agriculture (Oregon and Washington) established a policy dealing with soil productivity and erosion:

> . . . to plan and conduct land management activities so that soil loss from accelerated surface erosion and mass wasting caused by these activities will not result in an unacceptable reduction in soil productivity and water quality (Howes 1988).

Swanson and others (1989) provide an overview of the effects of erosion on long-term site productivity in the Pacific Northwest. However, their work stresses the results of studies in the Coast and Cascade Ranges in Washington and Oregon. The present discussion concentrates on erosion processes and resulting impacts on site productivity in the interior West, described in this symposium as the western-montane zone. The purpose of this paper is to: describe the potential effects of erosion on productivity, consider how the different erosion processes occurring on forest lands relate to these effects, and summarize the few available published reports documenting the magnitude of the reductions in site productivity associated with erosion in the western-montane region.

EROSION EFFECTS ON PRODUCTIVITY

The effects of erosion on site productivity result from a change in the total depth of soil material at a site or direct damage to vegetation. Changes in soil depth can affect productivity by changing the total nutrient pool and by changing the water storage capacity. Direct damage to vegetation is manifest by changes in mechanical support, changes in the availability of propagules, and direct damage to trees. I use the term "change" in describing the factors influencing site productivity because the reductions in productivity that occur at the site of erosion may be accompanied by increases in productivity at downslope deposition sites. This is especially true in forested settings where slope irregularities and large volumes of surface debris may cause deposition of eroded material within short distances downslope. For example, within the first 4 years after construction, over 95 percent of the material eroded from roadfills is deposited within 20 meters downslope from the bottom of the roadfill on granitic slopes in Idaho (Megahan 1984). Thus, it is important to recognize that the actual effects of erosion on productivity are represented by the net differences in productivity that occur at both erosion and deposition sites. The relative amount of soil loss and deposition depends on the type of erosion processes acting in the area and will be discussed in more detail later.

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Table 1—Mean horizon thickness, plant-available water, and nutrient content for four granitic soils in the Idaho Batholith (Clayton 1990)

Parameter	Horizon O	Horizon A	Horizon C
Thickness (cm)	3	25	46
Available water (percent by volume)	—	11.3	4.1
Range in K, Ca, Mg, N, P, S (percent of total)	82-93	4-11	1-7
Average K, Ca, Mg, N, P, S (percent of total)	88	9	3

The data in table 1 summarize the average total nutrient pool and plant-available water for four granitic soils in the Payette River drainage of Idaho. Note the large decreases in available water and nutrients with depth. The depth of erosion is important in terms of what is limiting at a site. If nutrients are limiting, loss of only 1.7 cm of surface organic matter will remove 50 percent of the total nutrients on the site. If water is limiting, as may often be the case in the western-montane area with hot, dry summers, all of the organic horizon and 17 cm of the A horizon must be eroded to remove 50 percent of the site's water-holding capacity.

Direct damage to trees caused by erosion occurs frequently in the interior West. Erosion may be severe enough to expose roots, reducing growth rates or reducing mechanical support to the point that the tree falls or is blown over by wind. Such damage can occur from erosion of surface soils (Carrara and Carroll 1979) but is especially prevalent on slopes adjacent to roadcuts due to rapid erosion of the steep roadcut surfaces (Megahan and others 1983). Tree fall caused by lateral erosion of roadcuts is a major concern for road engineers responsible for the maintenance of roads on steep slopes. Additional direct damage to vegetation is attributed to loss of propagules as surface soils are eroded. Seeds, root stock, and sometimes entire plants may be damaged or displaced by erosional processes. Finally, direct damage to large trees may occur from tilting, splitting, or abrasion of the trunks or by burying the lower portions of the trunk.

EROSION PROCESSES

The actual impacts of different erosion processes on site productivity are influenced by: (1) the depth of erosion (determines the amounts of soil components that are lost from the site), (2) the areal extent of the erosion (determines the area over which the losses occur), (3) the downslope rate of movement of eroded material, and (4) the probability of redeposition of the eroded material at downslope locations. Relative rankings of these four factors for each of the erosion processes described later are given in table 2. Evaluation of the total effects of erosion on site productivity must consider the net effects of all these factors.

Surface Erosion

Surface erosion is defined as the movement of individual soil particles by a force. Major factors regulating surface erosion include: soil cohesion, slope gradient, slope length, rainfall intensity, soil infiltration rate, and the amount of ground cover protecting the soil surface. Four different types of surface erosion processes are recognized and all are common in the Intermountain West. They are: splash, ravel, rills, and gullies. Splash erosion is caused by the impact of raindrops and occurs anywhere mineral soils are exposed. Splash erosion is most important on noncohesive soils. Ravel (sometimes called dry ravel or dry creep) occurs on steep slopes, generally over 60 percent, where gravity forces exceed the cohesive forces holding individual soil particles in place. It occurs during dry periods, primarily under the influence of wind, and is most common on noncohesive soils on bare roadfills and roadcuts and on natural slopes where logging, fire, or both, have exposed mineral soils. Rills and gullies (usually defined as rills more than 30 cm deep) are caused by channelized overland flow. Such flow is relatively rare on forest soils, even when bare, except where infiltration rates have been reduced by compaction, such as on skid trails or roads, or in the case of severe soil damage and the formation of water repellency as occurs on intensely burned areas.

Table 2—Properties of the different types of erosion processes

Erosion process		Depth of erosion	Areal extent	Rate of movement	Probability of slope storage
Surface erosion	Splash	mm-cm	widespread	m/yr	high
	Ravel	mm-cm	localized	m/yr-m/sec	high
	Rilling	cm	localized	m/day-m/sec	moderate
	Gullying	cm-m	concentrated	m/day-m/sec	low
Mass erosion	Creep	soil depth	widespread	mm/yr	high
	Earthflow	m-m x 10	localized	cm/yr-m/yr	moderately high
	Slump	m-m x 10	localized	m/yr-m/day	moderate
	Debris slide	cm-m	concentrated	m/sec	moderately low
	Debris flow	cm-m	concentrated	m/sec	low

In general, surface erosion rates are greatly influenced by the amount of vegetative cover and forest litter that are available for protection of the soil surface. Road construction and wildfires generally cause the greatest reductions in vegetative cover protection and thus commonly result in the greatest increase in erosion rates. However, even on roads and burned areas, erosion rates may decrease rapidly over time as revegetation occurs. Megahan (1974) found road erosion rates on granitic soils decreased about 90 percent by the second year after construction. Similar recovery was recorded about 2 years following a wildfire on a clearcut north slope in granitic soil (Megahan and Molitor 1975) but not following a controlled burn on a clearcut south slope in the same vicinity. In the latter case, considerable active erosion was still occurring 10 years after disturbance (Megahan 1990). Wildfire can cause greatly accelerated surface erosion. Connaughton (1935) evaluated the degree of accelerated surface erosion on an 18,000-ha wildfire in southern Idaho. Accelerated erosion was found on 42 percent of the area on cutover lands and on 28 percent of virgin forest land. In addition to the effects of logging on erosion severity, there were large increases in the severity of erosion with increasing hillslope gradient and burn intensity.

Of all the erosion processes, splash erosion is most widespread since it can occur anywhere bare soil is subjected to raindrop impact. However, the average depth of soil loss and the downslope rate of movement of eroded material are small. Thus, even though large volumes of soil may be moved by splash erosion, the total effect on forest site productivity is low. In contrast, gullying results in the rapid removal of considerable depths of material and transports that material long distances, usually to the nearest stream channel. In this case, productivity is greatly reduced by gully formation. Gullies normally occupy a very small area so the net reduction in productivity for the forest site is again small.

Mass Erosion

Mass erosion is defined as the movement of many soil particles *en masse*, primarily under the influence of gravity, and occurs when shear stresses exceed shear strength. Unlike surface erosion, which progresses from the surface downward, mass erosion usually includes the entire soil mantle and often part of the underlying parent material as well.

The five major kinds of mass erosion include: creep, earthflow, slump, debris slide, and debris flow (table 2). Creep involves imperceptibly slow (mm/yr) downslope movement of the soil mantle under the sustained influence of gravity on steep slopes. Effects on productivity are essentially nonexistent. Earthflow and oftentimes slumps tend to be deep-seated types of slope failures with the movement plane usually well beneath the soil in the underlying parent material or bedrock. Earthflows and slumps move meters to tens of meters per year and can cause direct damage to trees by tilting and splitting. Aside from road construction, effects of forest management activities on slumps and earthflows are not well defined.

Debris types of failures include debris slides and flows and involve the surface soil mantle sliding over the underlying bedrock or parent material. Lengths are long relative to their depth (by a factor of 20 times or more) and widths are generally small (meters to tens of meters). Debris failures occur on steep slopes usually greater than 60 percent. Slope depressions that serve as water accumulation zones are the most common sites for the initiation of debris failures. The release of the failures is sudden and movement rates are rapid with velocities of meters per second. Downslope delivery is relatively high, especially for debris flows, because of the high water content of the slide material. The primary factors affecting debris failures are: soil strength, vegetation roots, slope gradient, groundwater depth, and soil depth. Of these, vegetation roots, groundwater depths, and slope gradients (in the case of forest roads) are sensitive to forest management practices.

Debris failures have a severe impact on productivity at the site of failure because of the sudden, total loss of the soil mantle. Additional erosion sometimes occurs as a result of scour as the rapidly moving mass of eroded material moves downslope and from subsequent surface erosion in the slide site. However, the area affected is small because of the concentrated nature of the failures. Megahan and others (1978) collected data on 629 landslides in the Clearwater National Forest in Idaho. For a 3-year study period, the total area affected by landslides amounted to 16.5 ha, about 0.003 percent of the total nonwilderness area of the forest.

Wildfire can increase the number of debris failures per unit area of forest land. Jensen and Cole (1965) reported a total of 34,000 cubic meters of soil loss from the 400-ha Poverty Burn on steep slopes in the South Fork of the Salmon River as a result of debris failures. Assuming an average soil depth of 0.6 meters (reasonable for steep slopes in this area), this volume of soil loss would indicate that about 2 percent of the burn area was affected by debris failures. Although three orders of magnitude greater than the percentage of land affected by mass failures on the Clearwater National Forest, the loss of productivity occurred only on 2 percent of the Poverty Burn, a relatively small area.

Expressing productivity loss on the basis of area affected by debris avalanches can be misleading. This is because failure sites are usually located in slope depressions that serve as both soil and water accumulation zones. Because of greater soil depth and increased water availability, such sites also tend to be some of the better sites for tree production. Thus, total site productivity loss may greatly exceed the percentage of area affected by debris failures.

STUDIES DOCUMENTING EROSION EFFECTS

The effects of debris types of landslides where the entire soil mantle is lost all at once are relatively clear. Severe reductions in productive capacity occur until new soils accumulate at the slide sites. Smith and others (1986) reported a 70 percent reduction in conifer productivity

on slide sites in the first 60 years and about 50 percent at the end of 80 years on slides occurring on the Queen Charlotte Islands, British Columbia. A gradual increase in productivity is expected for subsequent rotations. No such data have been collected for slides occurring in the interior West. However, personal observations of slide scars in the region suggest that the magnitude and duration of productivity losses are at least as great as those reported by Smith and others (1986).

Aside from the formation of gullies, which would appear to have effects on productivity similar to those of debris failures, the effects of surface erosion processes on productivity are much more difficult to evaluate. On logged areas, other factors often associated with or causing erosion, including soil displacement and compaction by timber harvest equipment (Froehlich 1988) and the formation of water-repellent soil layers on burned areas (DeBano and others 1970), can also adversely affect productivity. Thus, studies of effects of timber harvest or fire on productivity have not clearly isolated the effects of surface erosion alone. Clayton and others (1987) showed that the degree of lateral soil displacement was associated with decreased tree diameter and height following logging of lodgepole pine (*Pinus contorta* Engelm.) and ponderosa pine (*Pinus ponderosa* Laws.) in central Idaho. Tree diameter (breast high) was reduced an average of 21 percent going from slight to high soil displacement; tree height was reduced an average of 24 percent. In this case, one might assume that the displacement effect would be a good indicator of erosion effects even though the displacement was not entirely caused by erosion. However, they also reported that tree diameter and height in the same area decreased an average of 19 and 18 percent, respectively, with increasing penetrometer resistance, an index of increasing compaction. Thus, it is impossible to discriminate between the negative effects of compaction and soil loss. Smith and Wass (1980) reported reductions in the height growth of Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) on logging skidroads on sensitive sites in interior British Columbia. But again, it was impossible to isolate the effects of erosion alone.

Because of confounding effects of other factors affecting productivity, it appears unlikely that it will be possible to accurately assess the effects of surface erosion on productivity based on field observations of tree growth in timber sale areas. Such studies are referred to as historical bioassay (HB) studies (Kimmins and others 1988). Also, HB approaches are based on observations for the growth conditions that existed during the life of the vegetation. Using the results of such studies for prediction purposes requires the assumption that the soil and atmospheric resources remain static. Such an assumption is open to question, especially at present when concerns for global climate change are widespread. Process models of vegetation growth processes offer an alternative for evaluating effects of erosion but have limitations primarily because of intensive input data requirements (Kimmins and others 1988). A hybrid simulation model called FORCYTE combines the best features of both HB and process model approaches while minimizing some of the problems. The latest version of the model (FORCYTE-11) appears to have promise for evaluating the effects of a number of components of the site-productivity question including the effects of erosion (Kimmins and others 1988). However, HB studies specifically designed to evaluate the effects of soil erosion alone are still needed to validate the model.

Klock (1982) developed an interesting alternative to the historical bioassay approach. He used a greenhouse bioassay technique to assess the effects of various amounts of erosion on four different forest soils in central Washington. Soil samples were collected from depths of 0-30 cm, 3-30 cm, 7.5-30 cm, and 15-30 cm to simulate respective erosion amounts of 0, 3, 7.5, and 15 cm. Growth of ponderosa pine, Douglas-fir, lodgepole pine, and orchard grass (*Dactylis glomerata* L.) in pots was used to index the effects of erosion. At the end of varying lengths of time, the vegetation was clipped at the soil level and ovendried and weighed. Erosion effects were evaluated by comparing the percentage of vegetation weight for the various eroded soils to that for the uneroded soil. Productivity losses ranged from none to as high as about 85 percent depending on the type of soil, the amount of erosion, and the type of vegetation. Klock (1982) concluded that, although the procedure does not provide a true measure of productivity loss from erosion, it does provide a means to compare the relative effects of different sites and erosion rates and to evaluate the sensitivity of different types of vegetation.

CONCLUSIONS

Erosion can cause large decreases in forest productivity at the site of soil loss. Reductions in productivity are directly, but not linearly, proportional to the depth of soil lost. The greatest and longest duration (decades) impacts are caused by debris landslides and gullies. Both landslides and to a lesser extent gullies are concentrated in small areas that tend to be soil and water accumulation zones. Such areas also tend to be relatively high in site productivity, so the percentage of loss of productivity exceeds the small percentage of the area affected by the erosion. Surface erosion removes much less total depth of soil than mass erosion, but may have a greater short-term impact on productivity because larger areas are affected. However, surface erosion rates tend to decrease rapidly over time (a few years) so the long-term effects are limited.

Erosion rates are minimal in the undisturbed forest but can be greatly accelerated by natural or man-caused disturbances. Of the various types of disturbances, wildfire has the largest potential for reducing productivity because of large soil losses over broad areas. Except for landslides and gullies, the erosional consequences of forest management activities are difficult to evaluate because of the confounding effects of other types of site disturbances. Considerable research, including both bioassay and process modeling techniques, is needed to better quantify the effects of erosion on site productivity. Studies need to consider the net effects of both on-site soil loss and downslope deposition of eroded material.

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THE EFFECT OF FIRE ON SOIL PROPERTIES

Leonard F. DeBano

ABSTRACT

Fire affects nutrient cycling and the physical, chemical, and biological properties of soils occupied by western-montane forests. Combustion of litter and soil organic matter (OM) increases the availability of some nutrients, although others are volatilized (for example, N, P, S). Soil OM loss also affects cation exchange capacity, organic chelation, aggregate stability, macro pore space, infiltration, and soil microorganisms. Nitrogen replenishment must be emphasized when prescribed burning programs are planned or during rehabilitation following wildfires.

INTRODUCTION

Fire significantly affects soil properties because organic matter (OM) located on, or near, the soil surface is rapidly combusted. The changes in OM, in turn, affect several chemical, physical, and microbiological properties of the underlying soil. Although some nutrients are volatilized and lost, most nutrients are made more available. Fire acts as a rapid mineralizing agent (St. John and Rundel 1976) that releases nutrients instantaneously as contrasted to natural decomposition processes, which may require years or, in some cases, decades.

The objectives of this paper are to (1) review the importance of OM and plant nutrients in the soil, (2) describe changes in OM during combustion, (3) characterize several soil physical, chemical, and biological temperature thresholds, and (4) identify and discuss the more important fire-related changes occurring in soils that affect postfire management.

ORGANIC MATTER

Organic matter in western-montane forest soils is concentrated on, or near, the soil surface and is made up of six easily recognized components: (1) the litter layer, consisting of recognizable plant litter; (2) the duff layer, composed of partially decomposed, but recognizable, plant litter; (3) the humus layer, consisting of extensively decayed and disintegrated organic materials, which are sometimes mixed with mineral soil; (4) decayed wood, consisting of the residual lignin matrix from decaying woody material that is on the soil surface or has been buried by the forest floor; (5) charcoal, or extensively charred wood mixed into the mineral soil; and (6) the upper mineral soil horizon (A horizon) of the underlying mineral soil (Harvey 1982). Soil OM plays an important role in the physical, chemical, and biological properties of the soil and, as such, contributes to overall soil and site productivity.

Organic matter acts as the primary reservoir for several nutrients and, therefore, is the source for most of the available phosphorus (P) and sulfur (S), and virtually all of the available nitrogen (N). Soil OM's role in N storage is especially important in forests because their continued high productivity depends, to a large extent, on large supplies of available N. Nutrients stored in OM are released slowly during decomposition, providing an efficient, steady source of nutrients that keeps leaching losses at low levels.

Soil OM and humus also provide chemically active cation exchange sites that retain many of the important cations (for example, NH_4^+, K^+, Ca^{++}). It has been estimated that soil OM can provide over 50 percent of the cation exchange capacity of some forest soils. It also is an active chelating agent that retains many of the metals.

Organic matter serves as a powerful aggregating agent and, as such, plays an important role in creating and maintaining a well-aggregated soil. Soil aggregation improves soil structure that creates macro pore space, and improves soil aeration. Aggregate soils also have higher infiltration rates than nonaggregated soils containing less OM.

The welfare of soil microorganisms also depends on OM because it provides both a suitable environment and C compounds that serve as an energy source for soil microorganisms. Both of these functions are critical for maintaining the nutritional quality and moisture-holding capacity of forest soils (Harvey and others 1987).

NITROGEN AND SULFUR

Nitrogen is an extremely important nutrient because it is the one that is most likely to limit tree growth in forests and other wildland ecosystems (Maars and others 1983). Because of this inherent limitation, significant losses of N during a fire could adversely affect long-term site productivity in many wildland ecosystems, particularly if N replenishment mechanisms are not provided for during postfire management.

Nitrogen contained in unburned forest litter and soil is released solely by biological processes and is referred to as being regulated by "biochemical cycling" (McGill and Cole 1981). Because of the close relationship between carbon (C) and N, C:N ratios play an important role in regulating the decomposition rate of OM and, as a result, control the rate at which N and other nutrients are released and cycled (Turner 1977).

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The role of S in ecosystem productivity is not well understood, although its fluctuations in the soil appear to parallel that of inorganic N. Sulfur is considered the second most limiting nutrient in some coastal forest soils of the Pacific Northwest, particularly when forest stands are fertilized with N (Barnett 1989).

PHOSPHORUS AND CATIONS

Phosphorus also has been reported to be limiting in some forest ecosystems. Deficiencies of P are most likely to appear in P-fixing soils (Vlamis and others 1955) or in conjunction with N fertilization applications (Heilman and Gessel 1963). Phosphorus uptake and availability to plants also appear to be highly dependent on the interrelationship between mycorrhizae and OM rather than being a simple absorption from the soil solution (Trappe and Bollen 1979).

Deficiencies of the major cations (Ca, Mg, and K) have not been reported for most wildland soils. The balance of these cations determines base saturation, which plays an important role in controlling pH regimes in soils.

COMBUSTION AND SOIL HEATING

During combustion, soil OM undergoes a series of physical and chemical transformations (Chandler and others 1983). Initially, the free moisture is vaporized as soon as the temperature approaches 100 °C. Lignin and hemicellulose begin to degrade at temperatures between 130 and 190 °C. Reactions occurring at temperatures below 200 °C are endothermic (reactions that require the absorption of heat). Decomposition of lignin and hemicellulose becomes rapid at 200 °C with cellulose undergoing chemical dehydration at 280 °C. About 35 percent of the total weight loss occurs before soil OM reaches 280 °C. Once soil temperatures exceed 280 °C, exothermic reactions (those reactions that produce heat) predominate and OM is ignited. When the surface temperature of soil OM reaches 500 to 600 °C, glowing combustion occurs if oxygen is not excluded from the char surface. Flaming then occurs and boosts temperatures from 800 to 1,500 °C. Above 1,000 °C carbon (C) is consumed at the surface as rapidly as char is produced.

Most of the thermal energy released during the combustion of aboveground fuels is lost upward into the atmosphere (DeBano 1974). However, a lesser but significant amount is radiated downward and is absorbed by the surface litter when present, or by other organic layers, depending on the amount and configuration of the OM deposited on the soil surface. The radiated heat can produce secondary combustion of the litter, duff, and, in some cases, the soil humus layer.

The amounts of litter, duff, and humus combusted depend on the duration and intensity of the heat flux reaching the litter layer. The combustion of aboveground fuels, and more important, duff, may heat the mineral soil surface significantly, and as a result substantial amounts of heat can be transferred downward into the soil by conduction, convection, and by vaporization and condensation. Thus, soil temperatures generated during fires vary considerably, depending on the fuel load and the burning conditions (DeBano 1989). If a large amount of fuel is present, soil temperatures can remain high for several hours and would be expected to produce large changes in soil chemical, physical, and biological properties. In contrast, soil temperatures produced during low-intensity fires used for fuel reduction may not produce appreciable changes in the soil OM.

CHANGES IN SOIL PROPERTIES IN RESPONSE TO HEATING

The spatial distribution of soil properties within a soil profile determine, to a large extent, the magnitude of change occurring in a particular soil property during a fire. For example, those soil properties located on, or near, the soil surface are more likely to be changed by fire because they are directly exposed to surface heating. As a result, organic material and related soil properties are more likely changed by radiated energy than other soil properties, such as clay content, which is often concentrated in subsurface layers where it is insulated from surface heating.

The sensitivity of a particular soil property to heating is also important. In general, changes in soil chemical properties are directly related to the changes in OM described earlier. However, some soil physical properties are also dependent on soil OM, while others are not (for example, clay content). Soil microorganisms are probably most sensitive to soil heating because they are living organisms that have relatively low lethal-temperature thresholds.

Soil Chemical Changes and Nutrient Losses and Availability

Nutrients contained in fuel and soil OM are cycled by biological decomposition processes in environments where temperatures rarely approach 38 °C and sufficient moisture is available for sustaining active microbial activity. Under these mild conditions, soil microorganisms decompose OM and slowly release many of the essential nutrients over time. In contrast, during a fire the nutrients stored in fuels and soil OM are subjected to severe heating and, as a result, undergo various irreversible transformations during combustion. However, the responses of individual nutrients differ and each has its inherent temperature threshold. Threshold temperatures are defined as those temperatures where volatilization of a nutrient occurs. For discussion purposes, these thresholds can be divided into three general nutrient categories: sensitive, moderately sensitive, and relatively insensitive. Nitrogen (Hosking 1938) and S (Tiedemann 1987) are considered sensitive because they have thresholds as low as 200 to 375 °C, respectively. Potassium (K) and P are moderately sensitive, having threshold temperatures of 774 °C (Raison and others 1985). Magnesium (Mg), calcium (Ca), and manganese (Mn) are relatively insensitive, with high threshold temperatures of 1,107 °C, 1,484 °C, and 1,962 °C,

respectively. Because the threshold temperatures of N, P, K, and S are lower than the flaming temperatures of woody fuels (1,100 °C and, except for P, lower than glowing combustion temperatures (650 °C), these nutrients are readily volatilized from OM during combustion.

Nutrient Losses—Because some nutrients, such as N, P, and S, have low temperature thresholds and are easily volatilized, it is important to consider their losses in more detail. Nitrogen will be used to illustrate nutrient losses by volatilization because it is the nutrient that is most likely to be limiting in forest ecosystems.

The amount of total N volatilized during combustion has been reported to be directly proportional to the amount of OM combusted (Raison and others 1985). Most of this volatilized N (up to 99 percent) reverts to N_2 (DeBell and Ralston 1970). This relationship may not hold at lower temperatures (Grier 1975), because OM can decompose without volatilizing N; therefore, N loss is not proportional to the loss of OM. The N that is not volatilized remains on the site either in uncombusted fuels or as highly available ammonium-N (NH_4-N) in the soil.

Phosphorus responds differently, and only about 60 percent of the total P is lost by nonparticulate transfer when fuels are totally consumed (Raison and others 1985). As a result, substantial amounts of highly available P can be found in the ash and on the soil surface immediately following fire. Percentage loss of S by volatilization is intermediate to N and P (Tiedemann 1987), and burning has been reported to remove 20 to 40 percent of the S in aboveground biomass (Barnett 1989).

Nutrient Availability—Most changes in nutrient availability result from two different processes: (1) in situ changes, and (2) translocation of organic substances downward into the soil.

Heating the underlying mineral soil directly affects nutrients contained in the soil OM (in situ changes). However, the responses of the different nutrients to heating indicate little change is likely to occur more than 4 to 5 cm below the soil surface, unless a very intense, long-duration fire occurs (for example, in piles of logs).

More important, nutrient availability (particularly N) in the soil can be increased by the translocation of nutrients downward into the soil during a fire. This occurs because steep temperature gradients are produced in the upper soil layers during the combustion of the litter and humus on the soil surface. During combustion, surface soil temperatures may exceed 1,000 °C. Poor heat conduction by the soil results in temperatures of 200 °C or less within 5 cm of the soil surface. As a result, some of the vaporized OM and ammonium-rich nitrogenous compounds released during combustion are transferred downward where they condense in the cooler underlying soil (DeBano and others 1976).

Although large amounts of total N are lost during the combustion of plants and litter, available NH_4-N is usually higher in the underlying soil following a fire because of the transfer mechanism (DeBano and others 1979). The increase in N availability (as NH_4-N) observed immediately following a fire appears related to the soil temperatures reached. For example, under an extremely hot fire most of the soil N is probably volatilized, particularly on or near the soil surface, and only small amounts are transferred downward in the soil. In contrast, under cooler soil-heating regimes, substantial amounts of NH_4-N can be found in the ash and underlying soil. Therefore, depending on the severity and duration of the fire, concentrations of NH_4-N may increase, decrease, or remain unchanged.

Phosphorus does not appear to be translocated downward in the soil profile as readily as N compounds. As a result, P increases mainly in the ash and on, or near, the soil surface (DeBano 1989; DeBano and Klopatek 1988).

Soil Physical Properties

Soil physical properties that are dependent on OM (for example, soil structure, pore space, aggregation) are all affected by heating during a fire. Other soil physical properties, such as clay content, are not readily affected, except on the immediate soil surface during a very intense fire. An important physical property affected by fire, one that regulates the hydrology of a soil, is water repellency (DeBano 1981). During fires, OM in the litter and upper mineral soil layers is volatilized (fig. 1A). Most of the volatilized OM is lost upward in the smoke, but a small amount moves downward along steep temperature gradients in the upper 5 cm of the soil and condenses to form a water-repellent layer that impedes infiltration (fig. 1B,C). The degree of water repellency formed depends on the steepness of temperature gradients near the soil surface, soil water content, and soil physical properties. For example, coarse-textured soils are more susceptible to heat-induced water repellency than fine-textured clay soils. The formation of this water-repellent layer, along with the loss of protective plant cover, increases surface runoff and erosion during the first rains following burning. A reduction in infiltration by a water-repellent layer can lead to extensive rill erosion on burned watersheds (Wells 1981).

Soil Microorganisms

Soil heating directly affects microorganisms by either killing them directly or altering their reproductive capabilities. Indirectly, soil heating alters OM (energy source) and increases nutrient availability, thereby affecting subsequent microbial growth. Although the relationship between soil heating and soil microbial populations is complex, it appears that duration of heating, maximum temperatures, and soil water all affect microbial responses (Dunn and others 1985). Microbial groups differ significantly in their sensitivity to temperature and nitrifying bacteria appear to be particularly sensitive to soil heating (Dunn and others 1985). Physiologically active populations of microorganisms in moist soil are more sensitive than dormant populations in dry soil.

Another important group of soil microorganisms that are particularly sensitive to soil heating during a fire are endo- and ectomycorrhizae. Because most ectomycorrhizae are concentrated in the OM on or near the soil surface, the loss of shallow organic layers may be at least partially responsible for the reported fire-related reductions

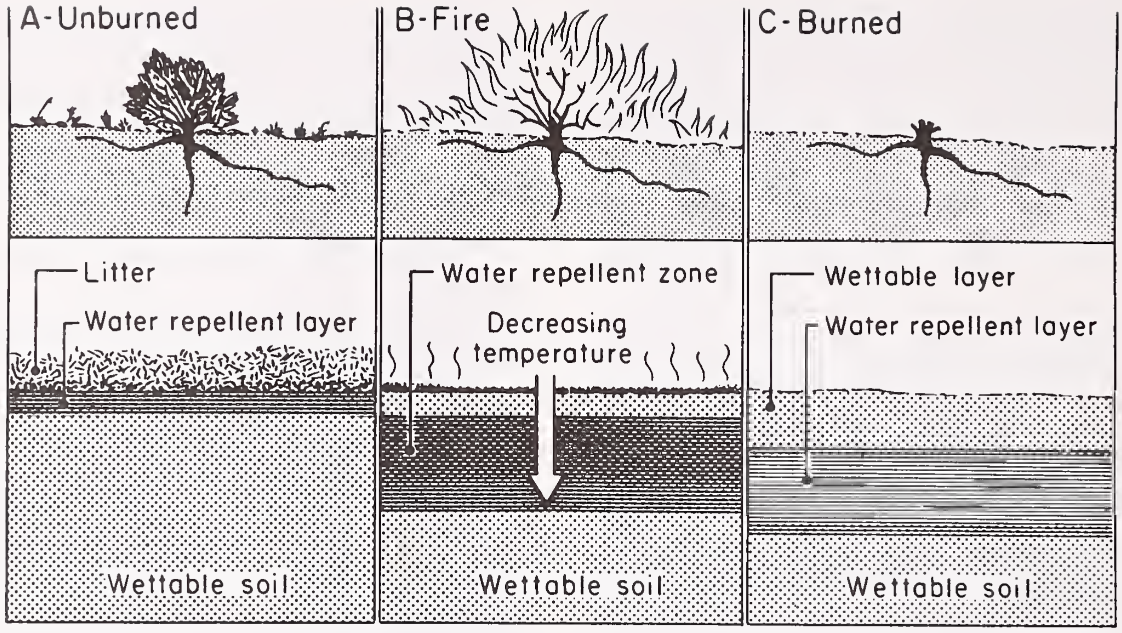

Figure 1—Soil-water repellency as altered by fire: (A) before fire, hydrophobic substances accumulate in the litter layer and mineral soil immediately beneath; (B) fire burns the vegetation and litter layer, causing hydrophobic substances to move downward along temperature gradients; (C) after fire, a water-repellent layer is present below and parallel to the soil surface on the burned area (DeBano 1981).

in ectomycorrhizal activity of western conifers (Harvey and others 1989). Likewise, ectomycorrhizae (for example, vesicular-arbuscular mycorrhizae [VAM]) in pinyon-juniper woodlands were also reported to be affected by soil heating (Klopatek and others 1988). This decrease in VAM colonization may be an important factor affecting the long-term productivity of forest ecosystems.

IMPORTANT MANAGEMENT CONCERNS FOLLOWING FIRE IN MONTANE-FOREST SOILS

Postfire management must play a key role in the development and implementation of any prescribed burning program or rehabilitation project following wildfires. Many of the general relationships between fire severity, soil heating, OM, and associated changes in soil properties discussed here have direct application when one assesses fire effects in soils in the western-montane forest environments and, thus, must be considered when postfire management guidelines are formulated.

Organic Matter

Although little can be done to control OM loss during wildfires, every opportunity must be taken to revegetate the site so that organic litter can again be restored on the site as quickly as possible. When one plans prescribed fires, more opportunities are available for maintaining an acceptable level of OM than occur following wildfires. For example, burning prescriptions can be designed to avoid burns that consume large amounts of surface litter and soil humus. Likewise, the total combustion of large woody debris on the soil surface (logs, etc.) during prescribed burning may not be a desirable practice. Repeated use of fire at frequent intervals probably should be avoided on relatively infertile sites where OM production is inherently low (for example, south-facing slopes), although it can play an important role in nutrient cycling in those ecosystems that experience frequent low-intensity fires (such as, ponderosa pine forests).

Nitrogen Loss and Replenishment

Because N is such an important nutrient in ecosystems and large losses are likely to occur by volatilization during the combustion of OM, special consideration must be given to both its loss and replenishment when planning burning programs. Important considerations to keep in mind when evaluating the effect of fire on N cycling are: size of the total N pool, type of fuel consumed, severity of the burn, and, more important, the mechanisms responsible for replacing N lost by volatilization.

Because of the large N losses, mechanisms for N replenishment in the soil must be considered an important part of postfire management. Nitrogen additions to the soil can come from several sources including: (1) small amounts of N present in precipitation and dust; (2) conversion or "fixation" of atmospheric N_2 gas into usable forms by soil- and root-inhabiting microorganisms; and (3) mineral or organic fertilizers. Atmospheric inputs are usually small in relation to other inputs and typically range from 1 to 4 kg/ha/yr in the Pacific Northwest

(Barnett 1989). Atmospheric N fixed by microorganisms ranges from 32 to 320 kg/ha/yr in fully stocked red alder (*Alnus rubra*) (Barnett 1989) to 0.1 to 1.3 kg/ha/yr by free-living N-fixers (Jurgensen and others 1979). Harvey and others (1989) found that more than one-third of the N-fixing capacity of some forest soils can be provided by microorganisms responsible for decaying wood on the surface and in the soil profile; thus, management of woody residues within a fire prescription may be an important dimension of N management in a fire environment.

CONCLUDING COMMENTS

Both wild and prescribed fires occur frequently in western-montane forests. These fires dramatically affect the nutrient cycling and the physical, chemical, and biological properties of the underlying soil. Substantial amounts of C, N, S, and P can also be lost to the atmosphere by volatilization during the combustion of litter, duff, and soil OM.

Because N is such an important nutrient in these ecosystems, the replenishment of N lost by volatilization during a fire must receive special consideration when burning programs are being planned or during rehabilitation following wildfires. Treatments interfering with the establishment of postfire N-fixing plants should be avoided, particularly on infertile soils having low site potentials. Woody residue management also appears to be an important factor in N fixation and may require special attention when fire prescriptions are being developed.

Burning increases the availability of most plant nutrients. Although some nutrients are volatilized during combustion, available NH_4-N and P increase substantially during burning. High concentrations of available plant nutrients on the soil surface immediately following fire may negate the advantage of fertilizing for at least 1 year after burning.

In the final analysis, fire plays an important role in the management of ecosystems, not only in the western-montane forests, but throughout the world. Although fire can dramatically affect soil properties and cycling, its effects can be mitigated by development of informed burning prescriptions or by careful selection of rehabilitation treatments following wildfires. However, careful planning is necessary to assure the sustained long-term productivity of these ecosystems is not adversely affected by fire-related changes in soils.

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SOIL PROCESSING OF AIR POLLUTANTS IN THE MONTANE WEST

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ABSTRACT

Western-montane forest lands have some of the cleanest air in North America, yet there are concerns about increasing levels of regionally dispersed ozone and acid deposition in alpine zones. Although SO_2 emissions are declining in much of the West, NO_x emissions are stable or increasing. Historically, the greatest air pollution damage has been caused by point sources associated with mining, smelting, and power generation. Such sources have caused extensive damage to local forests and forest soils.

INTRODUCTION

The western-montane forest region is defined here as forested land in the Western United States located east of the Cascade and Sierra Crests, and west of the Continental Divide, but including the Lewis and Gallatin Ranges in Montana and the Absaroka Range in Montana and Wyoming. This region is characterized as having some of the cleanest air in North America, although there is a history of localized air pollution damage to western forest stands resulting from metal smelting, refining, and electric power generation. The region is also periodically subject to pollution from large air masses from areas such as the greater Los Angeles Basin. Pollutants of concern include regionally dispersed ozone (O_3) and acid precursors (oxides of sulfur and nitrogen), and local sources of acid precursors, fluoride (F), and heavy metals arising from metal smelting and fossil fuel burning or processing. Some of these pollutants directly affect forest vegetation (O_3, F, acids). Metal pollutants are stored in soil and damage is mediated by soil biological and chemical processes. Acid deposition that reaches the forest floor is also processed by a variety of soil reactions.

Ozone

Ozone is the only regionally dispersed pollutant that has been demonstrated to injure foliage and cause forest decline or mortality in the West (Böhm 1989; Heck and others 1986), although I found no published reports of injuries to plants in the montane west. There is a perception by "experts" (18 to 35 North American scientists active in the field of forestry and air pollution) that volume yields of western conifers outside the Los Angeles airshed have probably declined slightly in the 1980's, and will continue to do so should current levels remain unchanged (de Steiguer and others 1990).

Adverse effects due to O_3 can start at ambient concentrations of 60 ppb. Bountiful, UT, has experienced average hourly O_3 concentrations in excess of 120 ppb during at least 6 of the 12 months (summer concentrations are usually higher), and the Lake Tahoe Basin has 12-month average concentrations exceeding 80 ppb (Böhm 1989). Ozone is at near-damaging levels virtually everywhere in the United States, but probably less so in the montane west (Fox 1990). Ponderosa and Jeffrey pines (*Pinus ponderosa*; *Pinus jeffreyi*) exhibit visual symptoms, reduced growth, and general decline in the San Bernardino Mountains, CA. Ozone damages chloroplasts, destroys chlorophyll, and reduces tree resistance to pests and pathogens.

Acid Deposition

Acid precursors (NO_x and SO_x) have not been detected in damaging concentrations in regional air masses in the montane west, although there are demonstrated vegetation declines downwind of point sources originating from ore smelting operations (Carlson 1990) and pulp and paper mill operations (Carlson and others 1974). Chemical concentrations of SO_x and NO_x in the west show a spatial pattern that is highly correlated with proximity to pollutant sources (fossil fuel refineries, pulp and paper mills, power plants for S, urban and industrial centers for N).

SO_2 emissions in the West for 1985 were estimated at 10 percent of the total U.S. emissions (Placet and Streets 1987). Major point sources are located in most western-montane States; the metal processing industry and electricity generation are the major polluters (Böhm in press). SO_2 emissions have generally decreased in the montane west over the last decade, although they increased by 10 percent in Wyoming and 25 percent in Idaho over this period (Böhm 1989).

NO_x emissions result primarily from the combustion of fossil fuels; consequently, emissions are largest around large cities and metropolitan areas. Emissions of NO_x remained fairly constant in the West in the 1980's, although States with new power plants (Arizona, Colorado, Montana, Nevada, and Wyoming) showed increases (Böhm in press). A gradual increase after 1990 is projected for the West as a whole (Young and others 1988).

There are numerous hypotheses to explain the role of acid deposition in forest decline in North America and Europe. These hypotheses include nutrient loss from soil (especially Mg, K, and Ca), mobilization of toxic elements (Al), and increased susceptibility to disease or insect attacks. The concept of "forest decline" is a category of diseases for which no

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single cause can be assigned. It is probable that forest damage from acid deposition generally results from multiple causes including changes in soil chemistry.

Particulates

Air pollution resulting from prescribed and wild fires is often discussed and publicized in local news media, particularly in the last few years because of the long fire seasons and large fires. Smoke from fires reduces visibility, increases complaints of respiratory ailments, and smells bad. The pollutant of concern is particulates emitted during the combustion process, and there is no evidence (or reason to expect) that particulates harm the forest ecosystem.

Fire effects are direct, on-site effects associated with damage to vegetation, loss of soil cover, and accelerated erosion, and are discussed elsewhere in these proceedings (see W. F. Megahan; L. F. DeBano). Clayton (1976) documented measurable down-wind transport of nutrients during a large wildfire in central Idaho, but this off-site effect, although measurable, was a small percentage of the annual nutrient requirement of the site. He concluded that redistribution of nutrients by smoke is ecologically unimportant.

Toxic Metals

Toxic metal air pollution is a common, local problem associated with smelting operations. With the exception of lead pollution from gasoline-powered vehicles, metal pollution is from stationary sources, and direct effects are typically within 10 km of the source. Lead is the only metal that currently has a national ambient air quality standard. There has been a dramatic nationwide improvement in ambient lead concentration from >1 $\mu g/m^3$ in the mid-1970's to <0.2 $\mu g/m^3$ in the mid-1980's (Council on Environmental Quality 1989). This improvement is a direct result of increasing use of unleaded gasoline and control equipment installed in stationary source stacks. Studies from the northeastern United States have not found that concentrations of metals in forest soils are correlated with regional patterns of deposition except near point sources (Friedland and others 1986); however, lead and cadmium concentrations in north-central U.S. forest floors are related to regional deposition patterns (Grigal and Ohmann 1989). Similar studies have not been conducted in the montane west, but the generally cleaner regional air masses and strong topographic effects on air movement make such associations unlikely. Litter decomposition as measured by microbial CO_2 evolution is repressed by some heavy metals (notably Pb and Zn), but not Cu, according to Moloney and others (1983). High levels of heavy metals in soils around smelters have been shown to decrease populations of soil microorganisms. Friedland and others (1986) suggested that metal concentrations ranging from 100 to 1,000 mg/kg are required to measurably decrease decomposition rates in the northeastern United States.

Mobility and bio-availability of metal cations following deposition generally decrease with time, increasing soil pH, and increasing clay content. Time allows for oxidation of metals, which decreases solubility, and time also allows for ion diffusion to strong sorptive sites or formation of relatively insoluble secondary mineral phases (Bohn and others 1979). The type of sorptive site controls how energetically, and, therefore, how persistently immobile an adsorbed cation will be. On inorganic exchange sites, competitive adsorption indicates that the relative replacing power corresponds to the order of increasing pH of the first hydrolysis product of the metal (for example, $Pb^{2+} + H_2O = PbOH^+ + H^+$) under acid conditions, but this is not the case on organic exchange sites. Elliot and others (1986) found that competitive adsorption followed the sequence Pb>Cu>Zn>Cd for mineral soils where organic carbon was <2 g/kg, but Pb>Cu>Cd>Zn for organic soils where organic carbon was >20 g/kg. Their results suggest that increasing soil organic matter will restrict the mobility of Cd and Cu.

Metal smelting usually results in increased air pollutant loads of nonmetals such as SO_2, acids, and fluoride (as HF or F^- salts). These accompanying pollutants are less persistent in soils, but often more directly damaging to vegetation on the site.

POLLUTION ASSOCIATED WITH MINING AND SMELTING

Northern Idaho and western Montana have a long and rich history of mining and smelting. For example, the Coeur d'Alene region of northern Idaho, which includes 10 mining districts, has yielded roughly 80 percent of the total value of all metals produced in Idaho since mining began in the State (about 1852). Exploitation of the great lead and silver lodes began in the 1880's, and the area is now as well known for the extensive pollution problems associated with the smelting as for the tremendous value of the processed ore. For example, the Bunker Hill lead and zinc smelters near Kellogg, ID, closed in 1982 after 70 years of continuous operation, are now a 54-km^2 Superfund site. The EPA has estimated that more than $100 million, expended over the next 50 years, will be required for cleanup. High emissions of SO_2 eradicated vegetation in the vicinity of smelters, and soils contain high levels of Pb, As, Cd, and Zn. Soil pH in the nonvegetated areas averages 3.1.

In 1955 an Al reduction plant was opened by Anaconda in Columbia Falls, MT. The process involves a high temperature, electrolytic reduction of Al_2O_3 to elemental Al, and results in particulate NaF and AlF_3 waste, plus gaseous HF. In 1970, daily emissions exceeded 3,500 kg of F, but these were reduced to less than 1,000 kg/day by mid-1971. Carlson and Dewey (1971) estimated that elevated F levels in vegetation could be found on over 80,000 ha of forested lands downwind of the plant. They also documented over 28,000 ha of forested land with visible injury to plants. Most F damage to vegetation is direct, and visual symptoms on conifers include needle necrosis and terminal bud dieback. Carlson and Dewey (1971) found a high incidence of injury on plants with >100 ppm F in needle tissue, and most plants with >30 ppm showed some degree of needle burn. They presented no data on soil F content. Normal concentrations of F in soils range up to a few hundred ppm. However, plant uptake of F is not usually related to F concentration in soil, but to other factors that affect availability, such as soil pH and calcium and phosphorus content of soil (Adriano and Doner 1982).

The damage resulting from copper smelting near Anaconda, MT, is even more extensive. The first copper smelter at Anaconda was built in 1884, and by 1914 was processing nearly 10,000 metric tons of ore daily (Carlson 1974). Accompanying this ore reduction were daily emissions of SO_2 estimated at greater than 2,500 tons! This caused extensive damage to the surrounding forest. Carlson (1974) cited historical evidence suggesting that nearly all conifers within 15 km of the smelter were dead by 1915. Since 1900 several abatement devices have been installed to reduce emissions, but the smelter was still emitting more than 1,000 metric tons of SO_2 per day in the early 1970's. The smelting operations were shut down in 1982. In 1989 the EPA claimed that Anaconda was its largest and most perplexing Superfund site.

In 1972 Carlson (1974) measured SO_2 and heavy metal deposition at soil plots located 7 to 18 km from the smelter. Plots located 7 km downwind of the smelter averaged 7 µg/cm^2/day of SO_2 deposition. Soil pH in the top 10 cm of mineral soil from these plots averaged 4.4 compared to a normal pH of 5.3 to 6.5. Lead content ranged from 50 to 60 ppm, and Zn content ranged from 130 to 200 ppm in the forest floor and top 10 cm of mineral soil. Carlson (1974) described "moderate" damage to live limber pine (*Pinus flexilis*), and he reported that prior vegetation damage resulted in sufficient mortality and ground cover loss to cause excessive erosion.

Walsh and Bissel (1979) investigated the effects of SO_2 and heavy metal emissions from the Anaconda Copper Smelter on two subalpine plant communities (*Abies lasiocarpa/Vaccinium scoparium* and *Abies lasiocarpa-Pinus albicaulis/Vaccinium scoparium* habitat types). They found dramatic changes in plant community composition and reduction in cover, which they attributed to air pollution. Overstory crown cover decreased from a mean value of 66 percent on distant control sites to less than 20 percent within 13 km of the smelter; shrub crown cover dropped from 50 to 66 percent at remote sites to 3 to 6 percent at 13 km. On the basis of pedestaled bunch grass, Walsh and Bissel (1979) estimated that accelerated erosion due to loss of ground cover had removed 15 to 30 cm of soil from one site located 11.5 km from the smelter. At this same site the authors found an absence of decomposed litter (Oa horizon), but 8- to 12-cm-deep layers of undecomposed needles (Oi horizon). They hypothesized that decomposer organisms may not be functioning properly due to air pollutants.

SOIL PROCESSING OF ACID DEPOSITION

Alpine and subalpine areas of the Western United States exhibit many characteristics that suggest that their aquatic ecosystems are vulnerable to acid deposition. These characteristics include: (1) shallow, coarse-textured soils, (2) a high percentage of the land area in rock outcrop, (3) soils with a low base saturation, (4) soils formed from rocks that contain minerals that weather slowly, and (5) rapid delivery of water to streams and lakes during summer snowmelt.

The higher elevation ecosystems of the Wind River Mountains in western Wyoming have all of these characteristics. The Wind River Mountains are also located 50 to 200 km downwind of SO_2 and H_2SO_4 sources that originate from coal-fueled electric power generation and natural gas fields that contain considerable H_2S. Clayton and others (in press a, b) described the soil distribution and characterized soil properties related to acid neutralization in the southern end of the range. Soils are formed from Precambrian acid igneous rocks, and are coarse-textured and not well developed (Inceptisols). Two soils, a Humic Cryaquept and a Dystric Cryochrept, plus large areas of rock outcrop, talus, and scree comprise the surface deposits in the basin. The Aquept is located along stream courses and lake margins. This soil contains 10 to 20 percent (by weight) organic matter in the A horizon, and has a relatively high cation exchange capacity (12 to 25 cmol$_c$/kg) and base saturation (0.4 to 0.98). The Ochrept is an upland soil, found in patches under a sparse canopy of whitebark pine (*Pinus albicaulis*), Engelmann spruce (*Picea engelmannii*), and subalpine fir (*Abies lasiocarpa*). This soil has a lower exchange capacity and base saturation than the Aquept. Both soils are capable of adsorbing SO_4 at ambient soil pH. This is an important mechanism for neutralizing H_2SO_4 deposition in older, well-developed soils such as Ultisols in southeastern United States.

Large, single-horizon soil columns were leached with H_2SO_4 in the laboratory to assess the capability of these soils to neutralize acid deposition (Clayton and others, in press b). The treatments were equivalent to 3 to 6 years of runoff and 50 to 100 times annual S deposition rates (estimated at 3 to 6 kg/ha/yr). A complete proton budget was computed for each leaching test, and the relative H^+ consumed by base exchange and mineral weathering, SO_4 adsorption, and $Al(OH)_3$ dissolution calculated.

Base cation exchange for protons was very effective at buffering all horizons during application of 12 to 16 pore volumes of pH 4 H_2SO_4. After 3 to 5 pore volumes of pH 3 acid, leachate pH dropped below 5 in the B and C horizons of the Ochrept. Sulfate adsorption and $Al(OH)_3$ dissolution became important neutralizing processes in those two horizons when pH dropped, and dissolved Al reached a concentration of 130 µmol/L in the C horizon. Over the entire experiment, cation exchange and weathering were the dominant neutralization processes, accounting for 56 to 96 percent of total proton consumption. Sulfate adsorption was important in the Ochrept horizons, accounting for 20 to 35 percent of total proton consumption. Sulfate adsorption was not important in the A and C horizons of the Aquept. This could be attributed to the abundance of organic matter, which inhibited SO_4 adsorption, or to the fact that pH did not decrease due to effective buffering by cation exchange. Postleaching analysis of the soils indicated that adsorbed Ca and K were approximately at the same value as unleached soil, although large amounts of Ca (up to 30 percent of exchangeable Ca) were detected in leachate. This suggests that hydrolysis of the abundant primary minerals in these youthful soils rapidly resupplies these cations under the conditions of the experiment. Clayton and others (in press b) concluded that the column leaching experiments indicated considerable capacity to neutralize acids in the soils studied, although they expressed concern for areas where meltwaters are not routed through soil before reaching a stream or lake.

CONCLUSIONS

The western-montane forest area is characterized as having some of the cleanest air in North America, yet it clearly has localized disaster areas resulting from small-plume air pollutants. These point sources are associated with smelting and refining activities, and stricter controls are improving this situation.

Regionally dispersed O_3 is increasing in the West, and it has the potential to damage forest ecosystems. Ozone's effect on trees is direct, and it is not processed by soils. There are concerns about the impact of acids and acid precursors in the West, particularly in alpine ecosystems. In general, SO_2 and H_2SO_4 emissions are declining in the West, but this is not the case with NO_x.

More study on the acid-neutralizing capacity of western alpine soils is needed, with particular emphasis on acid processing by rock and saprolite. We also know little about long-term time trends in soil properties and how soils might be affected by air pollutants. Johnson and others (1989) recommended soil archiving as a means of detecting gradual changes in soil properties as a result of human activities. This activity would be of particular benefit in the montane west where we suspect there are remnants of unpolluted soils.

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VEGETATIVE RESTORATION OF WESTERN-MONTANE FOREST SOILS

Richard Everett
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Phillip McColley

ABSTRACT

Understory aids in the restoration of physical, chemical, and biological soil parameters after forest disturbance. Understory establishment must be timely to reduce erosion and maintain forest processes; understory must occur in significant amounts over time to ameliorate disturbance effects, and it must meet management objectives. The goal of vegetative restoration of forest soils is to restore productivity potential to predisturbance levels in a time frame consistent with the cycle of disturbance.

INTRODUCTION

Understory (tree seedlings, shrubs, grasses, and forbs) contributes to the restoration of soil productivity after fire or tree harvest disturbance in western-montane forests. Understory vegetation establishment must be timely to reduce erosion and maintain soil processes and yet not exclude establishment of desirable tree species.

Vegetative restoration encompasses plant structures and processes that maintain or replenish important chemical, physical, and biological soil parameters impacted by disturbance. Early successional species are an integral part of the ecosystem recovery process toward a more stable steady-state condition following disturbance (Marks and Bormann 1972). Natural plant response may be adequate on some sites, but highly disturbed areas may require direct seedings or transplants to maintain soil productivity.

SOIL DISTURBANCE IN MONTANE FORESTS

Soil factors frequently affected by disturbance include organic matter content, nutrient content, bulk density and porosity, soil microflora and fauna, and microclimate. Forest disturbance also impacts nutrient cycling, erosion, and soil-development processes.

Increased erosion occurs after many types of site disturbance including fire, timber harvesting, residue treatments, and road building (Helvey and others 1985). Soil erosion may occur immediately after removal of the vegetative cover or several years later as roots from the prior vegetation decompose and no longer stabilize the soil (O'Laughlin 1974; Swanston 1974). Loss of the forest floor and surface mineral soil can be a serious obstacle to reforestation or rehabilitation of a site. Removal of the upper soil profile removes most soil organic matter, a large portion of soil nutrients, and beneficial soil biota such as mycorrhizae that are key components to soil productivity (Harvey and others 1980). Bioassay tests indicate significant declines in Douglas-fir (*Pseudotsuga menziesii* var. *glauca* [Beissn.] Franco) seedling growth and site productivity with the loss of surface soils, but results differ among soil types (fig. 1, Klock 1982).

Compaction of forest soils often is a problem after tractor harvesting and machine piling of the slash. Compaction causes loss of soil structure, porosity, aeration, and water infiltration. Reduced infiltration rates lead to increased overland flow and accelerated soil erosion. Compaction also limits root growth by reducing pore size, increasing mechanical resistance to root growth, and decreasing oxygen supply. Froehlich (1979) found up to 12 percent reduction in ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) growth rates after compaction.

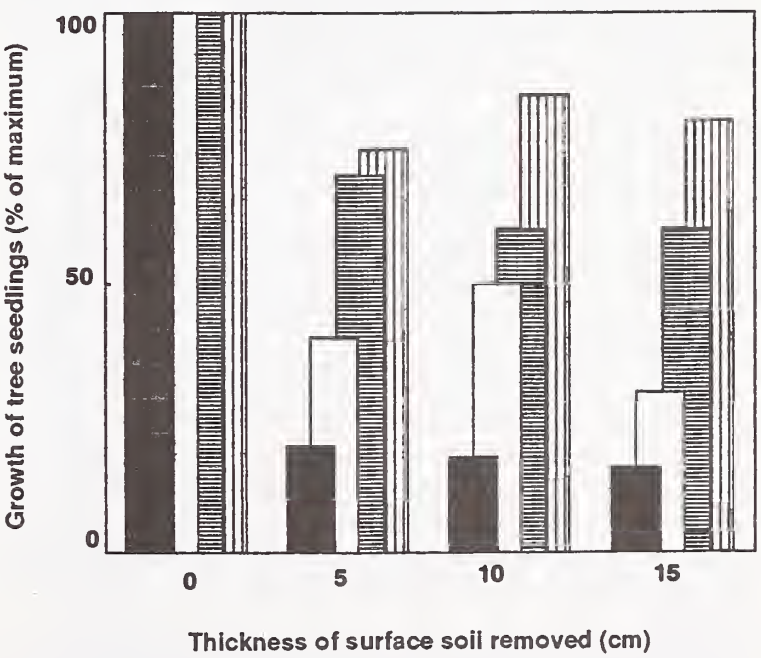

Figure 1—Growth of Douglas-fir seedlings in four eastern Washington soils after removal of successive surface soil layers (Klock 1982).

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Burning results in losses of organic matter, which include both soil organic matter and the aboveground forest biomass with its replenishing inputs of litter for developing soil organic matter. The loss of soil organic matter has detrimental physical, chemical, and biotic effects on soil productivity.

Fire not only eliminates much of the aboveground vegetation but also may eliminate large amounts of the forest floor (Grier 1975). Gessel and others (1973) report the forest floor of the east side of the Cascade Range contains 16,600 to 18,450 kilograms per hectare of organic matter and about 500 to 800 kilograms of nitrogen per hectare. In severe to extreme clearcut and burn treatments, Niehoff (1985) found a 22 to 83 percent reduction in organic matter and an 82 to 99 percent change in mineralizable nitrogen. Leaching losses can further reduce nutrient pools after disturbances, but these losses seem to be relatively small in this region (Clayton and Kennedy 1985; Hart and others 1980).

Many studies have shown that nutrient losses are a frequent result of various site disturbances in montane forests (Clayton 1981; Grier 1975). Removal of the vegetation reduces site nutrients directly through loss of the elements contained in the plant material. For timber harvesting, this can be a substantial amount of some elements (Clayton and Kennedy 1985). Disturbances from logging also can reduce nutrient supplies by erosion losses.

Soil microflora and microfauna populations are controlled by six environmental factors: moisture, temperature, aeration, pH, food substrates, and biological interactions (Harvey and others 1980). Changing any of these factors also can influence the other factors and cause changes in microbial populations (Bollen 1974). Fellin (1980) found clearcutting reduces soil mesofauna populations after harvest in western larch (*Larix occidentalis* Nutt.)/Douglas-fir stands. Fire generally reduces soil microorganism populations, but many populations increase after temporary reductions (Harvey and others 1989). Fire also can impact ectomycorrhizal fungi infection of root tips, but impacts from clearcutting may differ depending on the aboveground environment (Pilz and Perry 1984).

VEGETATIVE RESTORATION OF MONTANE SOILS

By definition, soil restoration occurs in the "reorganization phase" of stand development after disturbance (fig. 2). Here, forest processes fluctuate as the ecosystem attempts to reestablish steady-state conditions. Disturbance creates a declining cycle in forest processes or structure that is followed by an ascending cycle where parameters are increasing to predisturbance levels (Bormann and Likens 1979; Grier and others 1989; Klock 1982). The length and character of the reorganization and aggradation phases should vary with soil parameters, intensity of disturbance, and the activity of the restorative processes (including understory).

For forest floor biomass, the ascending phase may not occur until well after canopy closure occurs (Bormann and Likens 1979); erosion may lessen more rapidly as vegetation occupies the site (Helvey 1980). Vegetation can enhance soil nutrient status and soil processes by organic

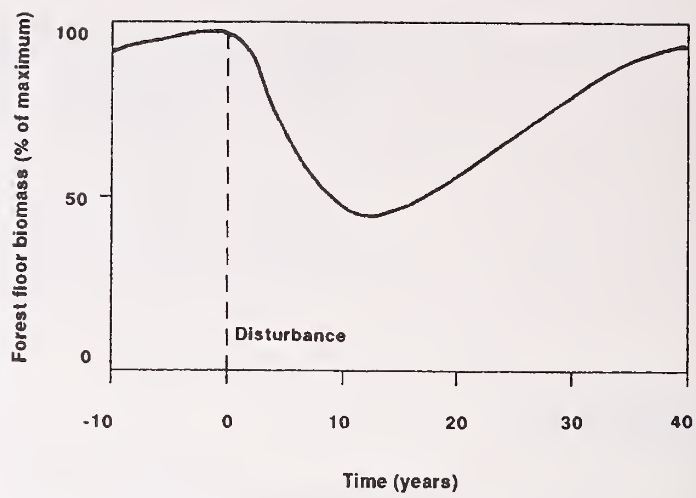

Figure 2—The hypothetical reorganization and aggradation phases for the forest floor biomass following tree harvest (adapted from Bormann and Likens 1979).

matter and nutrient inputs from throughfall, litterfall, root exudates, root turnover, N-fixation, and plant senescence and decay.

Vegetation rehabilitation also includes restoring or maintaining favorable soil structure, soil microbial processes, and a favorable microclimate. Although we discuss understory effects on soil restoration in this paper, we recognize that their inputs complement those of green tree retention, forest floor, and woody debris (Wollum and Davey 1978).

Timeliness of Establishing Vegetation

Timeliness of vegetation establishment is important in reducing erosion and leaching and in maintaining desirable soil processes (Perry and others 1989). Plants are effective agents for reducing erosion from raindrop impact, overland water flow, and wind (Waldron and Dakessian 1982). Rapid establishment of vegetation enhances capture of nutrients made available by disturbance and prevents their leaching off site (Klock and Grier 1979).

Perry and others (1989) present a good case for reducing intensity of disturbance to maintain understory species that preserve mutualistic soil-plant processes (N-fixation and mycorrhizae-nutrient cycling) from one forest rotation to the next, especially on harsh sites. These "bootstrapping" processes are theoretically maintained across rotations by understory-overstory "species guilds" that share common soil microbial associations. The reestablishment of these mutualistic processes can be difficult on harsh sites if guild species and associated symbionts are not rapidly established. What constitutes a guild member has not yet been defined, but the concept is a "nurse plant" for stand processes.

Duration of Understory Inputs

Duration of understory vegetation in the sere defines the "window" in which understory will make significant contributions to soil productivity. Shrub and herbaceous understory species reach maximum biomass and productivity in

early succession and then decline as tree dominance occurs. Snowbrush (*Ceanothus velutinus* Dougl.) can achieve site dominance within a decade in northeastern Oregon and have an aboveground biomass of 6,560 kilograms per hectare (McNabb and others 1979). Youngberg and Wollum (1976) report snowbrush produced 48,000-54,000 kilograms per hectare of aboveground biomass in a 10-year period. The importance of organic matter inputs from understory exceeds their relative biomass values, in that presence of the lesser vegetation with its litter can enhance microbial activity by faster decomposition and nutrient cycling rates (Youngberg 1965).

Many N-fixing understory species are the first to invade disturbed sites (Klemmedson 1979). Jurgensen and others (1980) identified five genera and seven species on nonleguminous N-fixing plants and eight genera with 25 species of leguminous plants present in the forests of Montana and central Idaho. Pioneer species from the genera of *Alnus*, *Lupinus*, *Ceanothus*, *Shepherdia*, and *Purshia* can contribute significant amounts of N during early stand development in specific habitat types, but are absent or underrepresented on many others (Jurgensen and others 1980).

The amount of N fixed depends on plant density, plant vigor, percentage of plants nodulated, and environmental conditions. Estimates of N-fixation for sitka alder (*Alnus sinuata* Reg. Rydb.) vary from 17 kilograms per hectare per year to 150 kilograms per hectare per year as shrub density increases from 1,830 to 5,380 and shrub age increases from 3 to 5 years (Heilman and Ekuan 1982). Another nonleguminous N-fixing species, snowbrush, is estimated to provide 72-108 kilograms of nitrogen per hectare per year in ponderosa pine and Douglas-fir stands, respectively (Youngberg and Wollum 1976). Low annual N-fixation rates for bitterbrush (0.57 kilograms of nitrogen per hectare per year) have been reported (Dalton and Zobel 1977). Although amounts of N fixed and released to the soil by snowbrush are under debate (Sylvester 1977), at least a portion of the N in litter (157-178 kilograms of nitrogen per hectare per year) and biomass (295-358 kilograms of nitrogen per hectare per year) (Youngberg and Wollum 1976) will become available to the developing stand at some point. Soils adjacent to snowbrush plants are N-enriched (50-75 percent greater total N) compared to soils where snowbrush does not occur (Binkley and Husted 1983).

Managing Stands and Species for Vegetative Soil Restoration

Jurgensen and others (1980) suggest snowbrush is the N-fixing species that offers the greatest potential for management. The proposed strategy is to use snowbrush to aid in conifer establishment and for soil N-enrichment followed by spraying to reduce snowbrush competition. Herbicide use is currently restricted, however, so that other alternatives to reduce competition are required. Hand-pulling of 1- to 2-year-old snowbrush seedlings is currently being practiced in the Wenatchee National Forest and is considered economically feasible at a cost of $119-180 per acre, according to Paul Schielke, Forest Silviculturist. Opportunities may exist to achieve a desired density that balances N-enrichment of the site with acceptable competition levels.

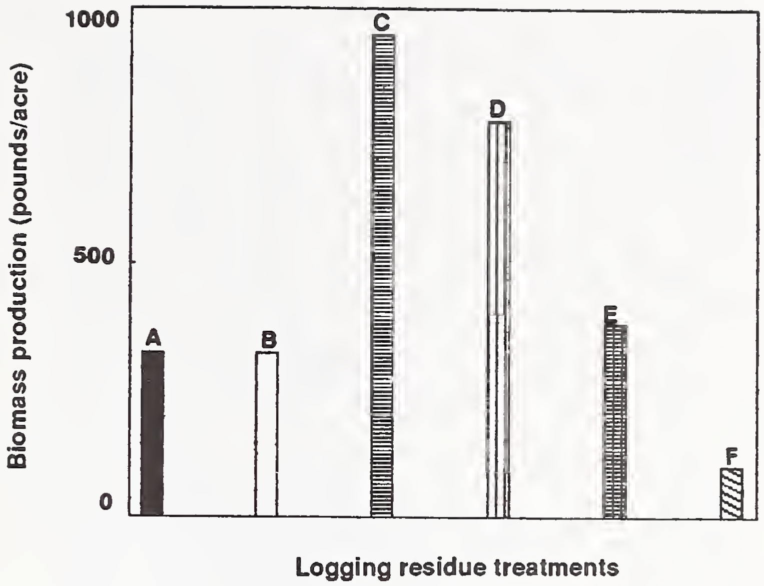

Figure 3—Understory biomass (pounds per acre) 6 years after tree harvest and residue management treatments in a lodgepole pine forest (Packer and Williams 1980). Residue treatments are (A) control; (B) chipped; (C) residue removed; (D) broadcast burned; (E) piled and burned in rows; and (F) piled and burned between rows.

We can manage for increased understory production and potential organic matter accumulation by silvicultural and residue treatments. Schmidt (1980) evaluated the effects of silviculture treatments on understory biomass in a larch and Douglas-fir forest. He found shrub biomass declined from pretreatment values in all treatments, but approached preharvest conditions as intensity of tree harvest increased. He found cover of herbaceous species and small shrubs increased over pretreatment values and were greatest in clearcut treatments. Understory biomass increased over controls when whole-tree harvest was practiced but declined when residues were chipped or burned in lodgepole pine (*Pinus contorta* Dougl. ex Loud) forests (Packer and Williams 1980; fig. 3). Understory response is species-specific for applied silvicultural treatments (Schmidt 1980).

Future vegetative restoration of soils may become difficult under proposed management strategies such as extended rotations and uneven-aged management that unduly prolong forest stands. Because many of our pioneer species fix N, we need to determine if the longer duration between their normal inputs will impact soil nutrient status. Wollum and others (1968) report the percentage of nodulated snowbrush plants declines exponentially with increasing time between snowbrush crops. Conard and others (1985) suggest this is a result of the absence of the endophyte caused by extended conifer domination of the site.

Establishing Understory Vegetation

Artificial vegetation establishment is warranted when natural response is inadequate to protect the soil resource or when native species are not as desirable. Both direct seedings and transplants have been used for vegetative restoration of disturbed forest soils. Seeding of grass species is recommended for rapid stabilization of disturbed

soils, especially by stream channels (Tiedemann and Klock 1976). Seeded grass cover on burned pine and fir forest soils significantly reduced the amount of bare ground in 1 year, but cover of natives was still inadequate by the fourth year on unseeded sites (Anderson and Brooks 1975).

Transplants are another way to rapidly establish plant cover on disturbed sites. Tiedemann and others (1976) were successful in establishing native shrubs on road cut and fill slopes in eastern Washington forests. Established bitterbrush (*Purshia tridentata* [Pursh] D.C.) and snowbrush may enhance natural succession because of their N-fixing capability. The reader is referred to seeding guides by Hafenrichter and others (1968) and McLean and Bawtree (1971) for species listings and their adaptability to forested sites. Although revegetation is a site-specific process, some general species considerations can be discussed. Grass species most often are used to control erosion on disturbed sites because of ease of establishment and rapid occupation of the site. Klock and others (1975) tested species adaptation of 40 grass, forb, and shrub species on severely disturbed fire lines in east-side Cascade forests. The success of most species decreased with increasing elevation (4,100 to 6,560 feet); this was attributed to freezing, frost heaving, short growing seasons, and high solar radiation levels. Poor performance of seeded grasses has been attributed to low soil N. Legumes have been included in the seed mix (Dyrness 1967) and starter N-S fertilizer has been applied with grass seed on east-side Cascade burn areas (Klock 1969).

Native and introduced legumes have been used in the interior forests as part of erosion control after logging or fire (Jurgensen and others 1980). Legume seedings have been more successful in spring than in fall (Anderson and Elliott 1957), but seeding failures are common (Anderson and Brooks 1975; Brooke and Holl 1988; Dyrness 1967).

Sites with compacted soils require species with roots capable of growth despite high mechanical resistance and poor aeration. Natural or artificial vegetation having this ability and able to provide organic matter to reduce bulk density and stabilize soil structure will be most successful. Alfalfa (*Medicago sativa* cv. Sonora) roots grew vigorously in compacted soils compared to grasses and ponderosa pine seedlings (Waldron and Dakessian 1982).

Introduction of species into natural ecosystems is an environmental concern in the maintenance of natural vegetative diversity. Helvey and Fowler (1979) found that seeded species reduce native plant cover by one-half (table 1) and adapted seeded species can persist for a decade or more (Geist 1976). Conversely, planting N-fixing species is suggested as a way to facilitate the invasion of native plants having higher soil nutrient requirements (Cargill and Chapin 1987). Although introduced grasses (orchardgrass [*Dactylis glomerata* L.]) may dominate early plant succession, the probability of their maintaining dominance declines rapidly with stand development, as on the Sleeping Child Burn in western Montana (Lyon 1976).

Table 1—Vegetative cover on sample plots located in a clearcut in the Blue Mountains of Oregon (Helvey and Fowler 1979)

Percent cover			
Seeded area		Unseeded area	
Native plants	Seeded plants	Native plants	Seeded plants
10.4	46.1	20.4	0.0

Soil Restoration and Conifer Competition

Vegetative restoration of soils and subsequent competition with tree seedlings remain controversial. Seeding less competitive grasses at 4-5 pounds per acre, followed by intensive grazing, has allowed conifer seedling establishment, but seedling growth was reduced in shelterwood cuts of eastern Oregon (Seidel and others 1990). The long-term benefits of reduced erosion and nutrient inputs must be balanced against reduced tree establishment and growth of tree seedlings that may extend for decades (Schmidt and Shearer 1973). Conversely, seedling establishment can be enhanced by shrub nurse plants (Youngberg 1965), especially on poor sites, in dry years, and with small planting stock (Conard and others 1985).

Both nutrient enrichment and reduced competition effects can be achieved if tree seedlings can be established before understory competition becomes severe (Zavitkovski and others 1969). Mechanisms need to be developed to reduce shrub competition once soil restoration has occurred.

CONCLUSIONS

Physical, chemical, and biological attributes of western-montane forest soils can be severely impacted by natural or prescribed fire and practices associated with tree harvesting and residue treatment. Establishment of vegetation is necessary to stabilize the soil surface and initiate processes that replenish nutrient levels and restore or maintain soil physical properties.

Timeliness of vegetation establishment is a factor in reducing erosion and nutrient losses, maintaining soil processes, and maximizing nutrient inputs. Dependency on unpredictable natural vegetation response can be ameliorated by direct seedings or transplants.

Vegetative restoration of soils must consider the maintenance of natural vegetation and safeguard future forest management options. We must also balance the short-term focus of limiting vegetative competition with the broader view of vegetative contributions to long-term site productivity. The defined ecological goal is the restoration or enhancement of soil conditions existing before disturbance in a time frame consistent with the disturbance cycle.

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ROOT DISEASE MANAGEMENT IN WESTERN-MONTANE FOREST SOILS

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ABSTRACT

This paper describes management of the following important forest root diseases associated with montane forest soils in interior Western North America: armillaria root disease, laminated root rot, annosus root disease, schweinitzii root and butt rot, tomentosus root rot, and black stain root disease. All these diseases are caused by fungi that spread primarily through soil via roots of woody plants. Disease management consists primarily of planting or favoring disease-tolerant conifer species on affected forest lands. Stump removal and chemical treatment have been used with some success, but more research is needed on potential detrimental effects. Models have been created to estimate and predict losses caused by some forest root diseases in western soils.

INTRODUCTION

Tree mortality caused by forest root diseases in the Western United States has been estimated at 240 million cubic feet (6.8 million m^3) of timber annually (Smith 1984). Root diseases are the most important pest problem facing current and future forests in the West, mainly because of the ability of disease-causing fungi to reside in forest soils on both living and dead woody tissue for decades. Some clones of root pathogens have been estimated to have been spreading for at least 460 years (Shaw and Roth 1976).

Six principal root diseases are caused by fungi in Western North America: (1) armillaria root disease caused by *Armillaria ostoyae* (formerly *A. mellea*), (2) laminated root rot caused by *Phellinus weirii* (formerly *Poria weirii*), (3) annosus root disease caused by *Heterobasidion annosum* (formerly *Fomes annosus*), (4) schweinitzii root and butt rot caused by *Phaeolus schweinitzii* (formerly *Polyporus schweinitzii*), (5) tomentosus root rot caused by *Inonotus tomentosus* (formerly *Polyporus tomentosus*), and (6) black stain root disease caused by *Leptographium wageneri* (formerly *Verticicladiella wageneri*). These diseases frequently occur together as complexes of two or more species of root pathogens and associated bark beetles (Goheen and Filip 1980; Hertert and others 1975). Abiotic agents such as drought, flooding, or soil compaction and other species of fungi may cause root disease, but by far, most damage is caused by these six principal species of fungi.

The biology of root disease fungi is similar. These fungi generally reside in the soil as associates of woody debris, primarily root material, rather than as free-living entities. Disease spread is chiefly through soil along the roots of woody plants. Common modes of spread are from dead woody material, especially stumps and dead trees, to roots of living trees. Spread from one living root system to another is common and especially damaging as inoculum loads increase. Root contact or grafting is not always necessary for root-to-root spread: some fungi such as *A. ostoyae* have specialized structures (rhizomorphs) that allow infection of roots in close proximity to one another.

Although all the root disease fungi form spores, the role of the spores in the soil is poorly understood. Except for *H. annosum* and *L. wageneri*, spread of root disease fungi by spores in the soil or through the air is thought to be negligible. Air-borne spores are the main mode of long-distance spread for *H. annosum*, and certain root-feeding insects transmit spores of *L. wageneri* from host to host. For a more detailed explanation of the biology of root disease fungi, refer to the following: for armillaria root disease (Shaw and Kile 1991), laminated root rot (Thies 1984), annosus root disease (Schmitt and others, in press), schweinitzii root and butt rot (Dubreuil 1981), tomentosus root rot (Whitney 1977), and black stain root disease (Harrington and Cobb 1988).

ROOT DISEASE MANAGEMENT

Use of root disease-tolerant species is the most common form of management of forest root diseases in Western North America. Removal of infected stumps and chemical treatments also have been used with some success, but little is known regarding potential detrimental effects of these treatments.

Tolerant Species

Degree of root disease tolerance varies with fungal and host species (table 1). Some root diseases seriously affect only certain host species: schweinitzii root and butt rot on Douglas-fir (*Pseudotsuga menziesii*), tomentosus root rot on spruce (*Picea* spp.), and black stain root disease on Douglas-fir and pines (*Pinus* spp.). Armillaria root disease, laminated root rot, and annosus root disease can cause severe mortality in several conifer species, especially grand fir (*Abies grandis*), white fir (*A. concolor*), mountain hemlock (*Tsuga mertensiana*), and Douglas-fir. Mortality also may vary by habitat type and level of stand disturbance (tree

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Table 1—Relative susceptibility of conifers in interior Oregon and Washington to damage by six common root diseases (adapted from Hadfield and others 1986)

Host species	Root diseases[1]					
	Armillaria	Laminated	Annosus	Schweinitzii	Tomentosus	Black stain
Douglas-fir (Rocky Mountain)	1	1	2	2	3	2
Ponderosa pine	2	3	2	3	3	2
Lodgepole pine	2	3	2	3	3	2
White pine	2	3	3	3	3	3
Sugar pine	2	3	3	3	3	3
Grand fir	1	1	1	3	3	4
White fir	1	1	1	3	3	4
Pacific silver fir	2	2	1	3	3	4
Noble fir	2	2	2	3	3	4
Subalpine fir	2	2	2	3	3	4
California red fir	2	2	2	3	3	4
Western hemlock	2	2	2	3	3	3
Mountain hemlock	2	1	1	3	3	3
Western larch	3	3	3	3	3	4
Engelmann spruce	2	2	3	3	2	3
Western redcedar	2	3	3	3	3	4
Incense cedar	3	4	3	3	3	4
Hardwoods	3	4	4	4	4	4

[1]1 = severely damaged;
2 = moderately damaged;
3 = seldom damaged;
4 = not damaged.

harvesting, road building, and so forth) especially for armillaria root disease (McDonald and others 1987). Tree mortality increases with degree of stand disturbance for black stain root disease (Goheen and Hansen 1978) and annosus root disease (Schmitt and others, in press).

Disease-tolerant and disease-resistant species (classes 3 and 4, table 1) can be favored during several silvicultural options including planting, seeding, using advance regeneration, precommercial thinning, commercial thinning, prescribed burning, seed tree and shelterwood systems, and group selection systems. If tolerant or resistant tree species are planted or regenerated for 50 or more years and ingrowth of more susceptible species (class 1 and 2) can be periodically removed, root disease fungi should die over most of the infected area. Subsequent rotations of susceptible species can be grown with little probability of reinfection. If tree species in damage class 2 are used, many trees will become infected but at levels less than if tree species in damage class 1 had been grown. Planting or favoring hardwood species has been tried, especially on sites affected by laminated root rot where red alder (*Alnus rubra*) has good potential to sanitize sites (Nelson and others 1978).

Stump Excavation

Often only highly susceptible tree species are regenerated on an infected site because of site conditions or economics. Maintaining tree vigor through precommercial thinning (Filip and others 1989) and fertilizing may minimize damage from armillaria root disease even with tree species in damage class 1. For some root diseases, precommercial thinning (Harrington and others 1983) or partial cutting (Filip and Goheen 1984; Hagle and Goheen 1988) may actually exacerbate root disease in highly susceptible species.

Theoretically, by removing the principal inoculum sources in infected areas, primarily infected stumps from harvested or dead trees, incidence of root disease on an infected site should decrease with time, even with highly susceptible tree species. Some studies have demonstrated this (Morrison and others 1988). The combining of stump removal with commercial thinning could provide (1) salvage of living but diseased trees, (2) root-free barriers limiting enlargement of the disease area, (3) destruction of root networks that support spread from scattered infections, (4) inoculum destruction, (5) elimination of fungal food bases, and (6) preparation of quality seed beds on some sites (Roth and others 1977). Some studies have shown increased growth of seedlings after stump excavation (Morrison and others 1988; Thies and Nelson 1988).

Whether stump excavation is effective in significantly reducing tree mortality caused by all root diseases on several soil types has yet to be demonstrated. Also, because stump excavation is typically accomplished with heavy machinary, the problem of soil compaction and soil layer mixing needs to be examined at several sites during different seasons. The benefits of retaining certain levels of woody debris in the soil (Harvey and others 1987; Maser and Trappe 1984) versus the detrimental effects of leaving root pathogen-infected material also need to be quantified. Stump excavation to control root diseases is seldom used in the interior West because of potential site impacts, uncertainty of results, and high costs.

Chemical Treatment

Application of fertilizer to manage forest root diseases has received only limited attention in Western North America. Low levels of certain soil nutrients have been associated with infection (Singh 1983) and decay (Shields and Hobbs 1979) caused by *Armillaria* sp. Others have shown that fertilizing with urea increases antagonistic soil fungi (Nelson 1975) and may increase tree resistance (Matson and Boone 1984) on sites affected by laminated root rot.

Fungicide application to protect stumps from root pathogen infection has been successfully demonstrated and is operationally used only for annosus root disease. Freshly cut stump surfaces are treated with a light coating of powdered borax to prevent germination of fungal spores (Graham 1971; Smith 1970). Use of fungicides to protect living trees from underground infection by *A. ostoyae* has been shown to be ineffective (Filip and Roth 1987).

Fumigation to eradicate *A. ostoyae* and *P. weirii* from infected stumps and even living trees shows more promise than protectant fungicides. Where stump excavation is impractical because of steep terrain or site and soil sensitivity, stump fumigation may be an alternative. *Armillaria ostoyae* can be eliminated from small infected stumps with such fumigants as chloropicrin, methyl bromide, carbon disulfide, Vapam, or Vorlex (Filip and Roth 1977). *Phellinus weirii* can be eliminated from infected stumps with chloropicrin, allyl alcohol, Vapam, or Vorlex (Thies and Nelson 1982, 1987). Use of fumigants to reduce mortality caused by root diseases in an entire infected area and potential detrimental effects of fumigation on nontarget organisms, such as soil fauna, mycorrhizae, and herbaceous vegetation, have yet to be demonstrated but work is under way (Thies, personal communication). Some experimental work to control laminated root rot with antagonistic soil fungi such as *Trichoderma* sp. in lieu of fumigants has been done with some success (Nelson and Thies 1985).

ROOT DISEASE MODELING

For the past 6 years, forest pathologists in Western North America have been developing a model to estimate and predict losses caused by root diseases (Stage and others 1990). This model is intended primarily as a tool for forest managers to quantify the benefits of applying various types of root disease management. When the western root disease model is linked to the Stand Prognosis Model (Wykoff and others 1982), more accurate tree growth and yields may be estimated for root disease-affected sites than if the Prognosis model is used alone. Currently only armillaria root disease and laminated root rot are modeled, but simulations have been attempted with annosus root disease (Shaw and others 1989), and formal modeling sessions are under way for that disease.

The western root disease model as linked with Prognosis can predict stand growth and yields following most of the standard silvicultural options: use of tolerant species through planting, natural regeneration, precommercial and commercial thinning, and seed tree and shelterwood harvests. In addition, stump excavation to remove inoculum with subsequent potential improvement in forest productivity can be simulated by the model. Increased accuracy in simulating stands affected by root disease will contribute to the forest planning process in the future.

CONCLUSIONS

Western-montane forest soils all too often contain one of six damaging root diseases and, frequently, more than one disease. Because of similarities in disease biology, there are management techniques and predictive models available to forest managers to reduce and quantify losses caused by one or more root diseases. The more commonly practiced management scenarios such as the use of disease-tolerant species will do little to detrimentally affect forest soils. Other management practices, however, such as stump excavation or fumigation, may decrease soil productivity. Additional research is needed to quantify these effects on several soil types throughout Western North America.

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Soil Information Management Problems

LEGAL PERSPECTIVES RELATED TO SOIL RESOURCE MANAGEMENT

Anne Puffer

ABSTRACT

Soil resource management and protection on forested lands, during forestry activities, is incorporated in several Federal acts. The Forest Service's soil resource management program reflects the requirements of these acts. Soil management and protection, during forestry activities, on State and private forest lands is most often incorporated in a State Forest Practices Act. In States without a Forest Practices Act, soil resource management and protection during forestry activities is voluntary. The Clean Water Act (CWA) provides a regulatory framework for soil resource management and protection on all forested lands. Soil that reaches and moves into stream channels is the major nonpoint source pollution from forestry activities on all forested lands. The CWA requires each State to identify and implement best management practices to reduce nonpoint source pollution on all lands.

INTRODUCTION

Federal and State laws and regulations exist that address the management and protection of individual natural resources (water, air, wildlife and fish, for example). At the present time, no specific Federal or State law exists that pertains to just soil resource management and protection.

Reference to soil resource management and protection on forested lands is found in several Federal laws and State Forest Practices Acts. The weakest link in legal direction related to soil resource management is in States, like Montana, Wyoming, and Utah, that do not have a Forest Practices Act. In States without acts, soil management and protection measures during forestry activities are voluntary.

FEDERAL LAWS

The Forest Service administers the majority of Federal forested lands. There are four principal Federal laws relating to the Forest Service that include provisions for soil resource management/protection. They are the Organic Administration Act of 1897, Multiple-Use Sustained-Yield Act of 1960, Forest and Rangeland Renewable Resources Planning Act of 1974, and National Forest Management Act of 1976.

In the Organic Act, one of the initial purposes for establishing any unit of the National Forest System (NFS) is "to improve and protect the forest within the boundaries . . .." The Multiple-Use Sustained-Yield Act set forth the secondary purposes of establishment "for outdoor recreation, range, timber, watershed, and wildlife and fish purposes."

The Forest and Rangeland Renewable Resources Planning Act (RPA) requires an assessment of the present and potential productivity of the land. It also requires that the Forest Service develop a program, in full accord with multiple-use and sustained-yield principles, that includes recommendations which ". . . recognize the fundamental need to protect and, where appropriate, improve the quality of the soil, water, and air resource." The act directs the Forest Service to develop regulations that set forth the process for development and revision of land management plans. The regulations are required to include as a minimum guidelines that:

> . . . provide for obtaining inventory data on soil and water, including maps, graphic material, and explanatory aids; . . .
>
> . . . provide for methods to identify special conditions or situations involving hazards to the various resources and their relationship to alternative activities. . . .

Additionally, the regulations are to specify guidelines for land management plans developed to achieve the goals of the program that:

> . . . insure that timber will be harvested from NFS land only where . . . soil, slope, or other watershed conditions will not be irreversibly damaged.

The Forest Service planning regulations (36 CFR 219.14, 219.23, 219.27) incorporate the RPA requirements for conserving the soil resources to maintain soil productivity.

The National Forest Management Act amended RPA by adding sections that stressed the maintenance of productivity and need to protect and improve the soil and water resources, and avoidance of permanent impairment of the productive capability of the land. The act also amended Section 18 of the Knutson-Vandenburg (KV) Act to authorize the use of KV funds to protect and improve the future productivity of the renewable resources of the National Forests, including soil and water.

WATERSHED PROGRAM

The primary objective of the Forest Service's watershed program is: "To protect and, where appropriate, enhance soil productivity, water quality and quantity, and timing

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of water flows." The policy of the Forest Service is to "design all management activities of other resources to minimize short-term impacts on the soil and water resources and to maintain or enhance long-term productivity, water quality, and water quantity."

The Forest Service carries out watershed planning to identify and evaluate watershed conditions and plan appropriate corrective action as needed. The policy of the Forest Service with regard to watershed planning is that the protection and development of soil and water resources will be components of land and resource management planning for NFS land. Soil capabilities and land-base limitations are to be identified in all land and resource management plans. Enough information must be available to design management alternatives that will protect and, as appropriate, improve the quality of the soil resource.

To protect watersheds, the Forest Service evaluates watershed conditions on NFS lands to (1) assess long-term trends of soil and water conditions as influenced by integrated land uses practices; (2) assess the changes in resource outputs that result from changes in watershed condition; and (3) effectively carry out land management activities without permanent impairment of soil productivity.

Work associated with resource improvement practices implemented to improve watershed condition is accomplished through watershed improvement projects. The objective of the projects is to restore the hydrologic balance of degraded watershed areas by stabilizing soil, controlling surface runoff and erosion, and improving long-term soil productivity.

FOREST PRACTICES ACTS

Forestry activities on State and private lands are most often regulated through a State Forest Practices Act. Half of the western States have a Forest Practices Act (FPA). The FPA's recognize that State and private lands contribute social and economic benefits to the State by helping to maintain healthy forests, soil, air and water resources, and providing habitat for wildlife and aquatic life. Most also recognize the need for a balance between the implementation of forest practices and the protection of forest soil, air, and water resources.

The FPA's are implemented through State regulations or rules that establish the minimum performance standards for the conduct of forest practices on forest lands. In general the following criteria are applicable to all of the regulations:

- Provide for the harvesting of forest tree species in a manner that will maintain the productivity of the forest land, minimize soil and debris entering streams, and protect wildlife and fish.

- Provide for road construction that will ensure protection and maintenance of forest productivity, water quality, and fish and wildlife habitat during construction and maintenance.

- Provide for reforestation that will maintain a continuous growing and harvesting of forest tree species by describing the conditions under which reforestation will be required, requiring stabilization of soils that have become exposed as a result of harvesting.

Forest Practices Act regulations are generally not highly prescriptive with regard to how forest activities must be carried out. In most cases they are performance standards that specify the level of resource protection, not prescriptive standards that specify the exact measures to be used to attain a given level of resource protection. The basic reason behind this is that site conditions vary widely across a State's forestlands, therefore it is not possible to establish a statewide set of prescriptive rules that would be applicable to all forest activity sites.

Examples of standards for the conduct of forest practices in the regulations implementing FPA's are:

- Select for each harvesting operation the logging method and type of equipment adapted to the given slope, landscape, and soil properties in order to minimize soil erosion.

- Limit the grade of constructed skid trails on geologically unstable, saturated, or highly erodible or easily compacted soils to a maximum of 30 percent.

- Locate landings on stable areas to prevent the risk of material entering streams. Avoid excessive excavation and filling.

- During landing cleanup, where exposed soil is unstable or erodible and may be reasonably expected to cause damage to a public resource, it shall be seeded with grass, clover, or ground cover or compacted, riprapped, water barred, benched or mulched, or be treated by other approved means.

- Where feasible, do not locate roads on excessively steep or unstable slopes or known slide-prone areas. Slope stability will be determined using available soils information, or evidence of geologically recent slumps, or where the natural slope exceeds the angle of repose for the particular soil types present.

- Road construction shall be accomplished when moisture and soil conditions are not likely to result in excessive erosion or soil movement, so as to avoid damage to public resources.

CLEAN WATER ACT

In an indirect way, the Clean Water Act (CWA) is a regulatory framework for soil management on all forested lands. Section 319 requires each State to develop a nonpoint source (NPS) management program plan. The NPS program plan is required to identify best management practices (BMP's), which will be used to reduce NPS pollution loadings by NPS category. One of the categories is forestry. Soil that reaches and moves into stream channels is the major NPS pollution from forestry activities on all forested lands. Therefore, soil resource information, such as erosion hazard, is essential in the site-specific design of BMP's to reduce NPS pollution on forest lands.

The NPS program plan is also required to identify programs (either regulatory or nonregulatory) to achieve BMP implementation. The water quality standards for the States of Idaho and Montana, developed per Section 303 of the CWA, are examples of regulatory programs for controlling NPS pollution. In Idaho, the rules and regulations pertaining to the Idaho FPA are specifically identified in

the State's water quality standards as approved BMP's for the State. The Montana water quality standards require the implementation of all reasonable soil and water conservation practices (analogous to BMP's) to control nonpoint source pollution. Although Montana does not have a FPA, it has developed BMP's for forestry. While considered to be voluntary, they are viewed as the minimum performance standards for all forestry activities in the State. Because implementation of BMP's is specifically addressed in both State's standards, forestry activities on Federal, State, and private forest lands must meet the intent of the water quality provisions of the approved forestry BMP's.

On National Forest System lands, the soil and water conservation practices described in the joint Intermountain and Northern Region Forest Service Handbook 2509.22 (Soil and Water Conservation Practice Handbook) are the vehicle by which State water quality protection provisions of the BMP's are implemented during forestry activities.

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USING SOILS DATA TO DEVELOP A NATIONAL FOREST MANAGEMENT PLAN

Gordon E. Warrington

ABSTRACT

The final Bridger-Teton National Forest Plan was built on the concept of Tentatively Suited lands for management. Attribute data about soils, slope, and vegetation were processed by a geographic information system to display the location of these lands and provide information about other controllable management elements. Management alternatives, based on land characteristics and desired future conditions, were mapped to communicate forest resource allocations that meet different sets of social values.

INTRODUCTION

Allocating National Forest resources to meet society's social values (Kennedy 1985, 1988) is the goal of management planning. This planning process is aimed at providing a satisfactory amount and mix of resource outputs while protecting use options for future generations. Forest Service regulations direct that management be planned for implementation on suited lands. Tests for management suitability of lands are based on currently available technology to ensure that irreversible resource damage to soil productivity or watershed condition will not occur (Congressional Record 1976; Federal Register 1982).

This paper outlines the process used by Bridger-Teton National Forest (BTNF) personnel to develop management information for the final Forest Plan. Soil characteristics along with land slope and vegetation formed the foundation for this land stratification to identify Tentatively Suited lands. A geographic information system (GIS) was used to systematically identify and display the location and suitability characteristics of BTNF lands.

MANAGEMENT INFORMATION

Managing forest operations to achieve desired outputs involves planning, organizing, coordinating, directing, controlling, and supervising. These management activities are usually successful when appropriate information is communicated about particular facts or circumstances that are relevant to natural resource management processes.

Kennedy (1985) points out that natural resource managers respond to four major interrelated systems. These are represented by economics of utilizing land, labor, and capital resources; social values, cultural attitudes, and behavior; political and legally defined policies; and environmental/natural resource biosphere components and processes. I would add one more system, technological capabilities to process data and carry out tasks.

A natural resource manager's only method of responding to social, environmental, and technological systems is to manipulate a discrete set of management elements. Therefore, the key to managing natural resources is knowing what can be controlled. Controllable management elements are those variables that a manager can manipulate through a decision to achieve the corporate goals and objectives. For natural resource management the controllable elements are (Warrington 1989; Warrington and others 1990):

- Quality is the goal toward which the methods used to implement the management practices are aimed (Pirsig 1974). It is expressed through the effects of the chosen management practices on the functioning and productivity of affected watersheds. This includes the extent of disturbances, the magnitude of disturbance, and the duration of the effects of the disturbance.

- Quantity of outputs produced (for example, board feet, animal unit months [AUM's]) or inputs used (such as tree planting, range improvements).

- Location of the practices on the ground.

- Timing of practices through the sequencing of entries into a watershed and/or the season of operation. If only one management entry is made then only the operating season is important. With multiple entries the operating season has an effect on the watershed response for each entry as well as on the cumulative effects of all entries.

- Mix of outputs can be changed through crop rotations. This element is more important in agriculture than in forestry or range management because crops can be changed in shorter time periods than are practical for forests or rangelands.

These controllable management elements are the basis for communicating management information. Management information is created by upward aggregation of data and interpretations from several sources (fig. 1). The bottom of the pyramid represents basic data (individual facts) collected during inventory work. This is usually a large data set, but without further characterization it does little to explain observed phenomena. Therefore, basic data are interpreted to provide estimates about potential outcomes based on appropriate sets of inputs. Interpreted information is modified by social and economic constraints to become management information.

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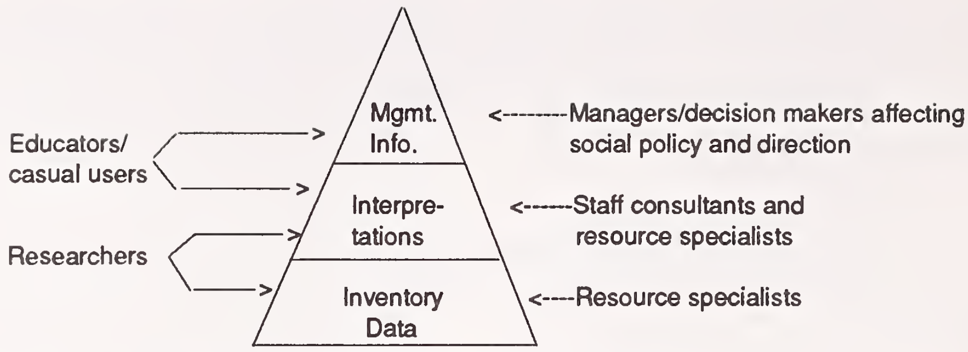

Figure 1—Schematic representation of natural resource user groups and their general information needs.

Six broad categories of natural resource information users are identified in relationship to their informational needs. The kinds of information and categories of users are not necessarily distinct. A researcher may also be an educator, and so forth. These groupings do, however, provide a basis for making inventory data into a useful management tool for forest planning.

TENTATIVELY SUITED LANDS

The first step in allocating BTNF resources was to assign all forest lands that were available for resource manipulation to two categories of management suitability (USDA Forest Service 1989). These categories, based on watershed resource characteristics, are Tentatively Suited and Not Suited for forest management practices. These management practices include but are not limited to:

• Timber production, including harvest, site preparation, and planting.

• Timber Yarding when moving a log from the stump to a landing.

• Roads, including construction, maintenance, closure, and restoration to natural contours. The potential for resource damage to areas outside of the road right-of-way is also considered.

• Surface Occupancy, including construction activities and continuous onsite activities.

Slope gradient and slope stability were the primary factors that best represented the overall management suitability of BTNF lands. The Tentatively Suited and Not Suited land areas were identified and mapped from interpretations of soil stability characteristics and land slope. For combinations of slope stability and gradient where technology is not available to prevent irreversible damage to soil productivity and watershed condition, the land areas are classified as Not Suited for management in the foreseeable future. Due to the inherent variation in landscapes and mapping standards, small areas of Tentatively Suited land may not be delineated, and some other lands may be mapped as Not Suited. This makes onsite verification of management suitability a requirement for all projects.

Land suitability categories (USDA Forest Service 1989) based on land slope and land attributes are shown in figure 2. Not Suited lands fall into the combinations of attributes shown to the lower right of the asterisks dividing the matrix. As land capabilities change, the cost of managing Tentatively Suited lands also changes. An economic and social cost analysis will be used to determine management feasibility on those lands.

LAND SUITABILITY CRITERIA

Following are short explanations about each of the soil and slope criteria used for stratifying land suitability. A complete discussion can be found in USDA Forest Service (1989).

Slope Classes

<=40 Percent—This seems to be a generally accepted slope break based on local experience where land management practices have shown extensive damage to occur when equipment was operated on slopes over 40 percent.

41-55 Percent—Resource damage along mountain roads is often related to the height and stability of road-cut banks. Therefore, a finished 12-foot-wide road with a ditch will require an overall width of 16 to 17 feet. Using a balanced cut and fill design, a $1^1/_2$:1 cut-and-fill slope, and a vertical cut height of 30 feet, the resulting land slope is approximately 55 percent (USDA FS and USDI BLM 1976).

56-70 Percent—Changing road design to a full bench on slopes >55 percent avoids long, unstable sliver fills that are difficult or impossible to compact. In the Oregon Coast Range, Sidle and others (1985) reported that the number of road-related landslides has been reduced by using full-bench construction on slopes >26° (49 percent).

>70 Percent—The slope is generally steeper than the angle of repose for unconsolidated natural materials.

Stability Classes

Each soil-map unit is rated for its risk of failure (USDA Forest Service 1976; USDA Forest Service and USDA Soil Conservation Service 1986) using one of four hazard levels. This rating is based on land characteristics that indicate potential for mass failures along with frequency of actual landslides delineated in the Geological Hazard Inventory.

Soil/land attribute	Slope class 1 0-40 percent	Slope class 2 41-55 percent	Slope class 3 56-70 percent	Slope class 4 >70 percent
1 Stable	TH=Yes Y=Conventional R=Yes SO=Yes	TH=Yes Y=Cable R=Yes[1] SO=Yes[1]	TH=Yes Y=Cable R=Yes[2] SO=[2]	TH=Yes[1] Y=Aerial R=Yes[2] SO=None
2 Marginally stable	TH=Yes Y=Conventional R=Yes SO=Yes	TH=Yes Y=Cable R=Yes[1] SO=Yes[1]	TH=Yes Y=Cable R=Yes[2] SO=[2]	TH=Yes[1] Y=Aerial R=Yes[2] SO=None
3 Marginally unstable	TH=Yes[1] Y=LI R=Yes[1] SO=[1]	TH=Yes[1] Y=Skyline R=Yes[2] SO=None	TH=No Y=None R=None SO=None	TH=No Y=None R=None SO=None
4 Unstable	TH=Yes[2] Y=LI R=Yes[2] SO=[2]	TH=No Y=None R=None SO=None	TH=No Y=None R=None SO=None	TH=No Y=None R=None SO=None
5 Landslide	TH=Yes[2] Y=LI R=Yes[2] SO=[2]	TH=No Y=None R=None SO=None	TH=No Y=None R=None SO=None	TH=No Y=None R=None SO=None
Alpine cirque basins and slopes with snow avalanche paths	TH=No Y=None R=None SO=None	TH=No Y=None R=None SO=None	TH=No Y=None R=None SO=None	TH=No Y=None R=None SO=None

Figure 2—Bridger-Teton National Forest land suitability matrix. Symbols and relative amounts of mitigation needed to maintain acceptable watershed condition: TH = Timber Harvest Method, [1]Some Restrictions, [2]Many Restrictions; Y = Yarding Method, LI - Low Impact; R = Roading, [1]Some Restrictions, [2]Many Restrictions; SO = Surface Occupancy, [1]Some Restrictions, [2]Many Restrictions.

A mass failure hazard rating of Stable indicates that evidence of past mass movement is not discernible and land characteristics are not conducive to future mass movement. A Marginally Stable rating indicates that evidence of past mass movement has not been discerned, but there are land characteristics that are conducive to mass movement. A Marginally Unstable rating indicates that evidence of past mass movement exists, but no current movement is discernible. An Unstable rating indicates that the site is actively moving, and probabilities of increased or additional movement, even without human-caused disturbances, are high.

Landslides

Geologic-hazard (landslide) mapping (DeGraff and others 1979) was conducted by the Forest Service Regional Environmental Geologist and contractors. Both recently active landslides and landslides dormant since recession of the last glacial period were delineated on color aerial photography (1:15,840) and transferred to a 1:24,000 orthoquad base.

The dormant landslides may be activated, particularly with changes resulting from road building, timber harvest, and burning practices. In general, the greatest potential for new landslides occurs in areas with a history of past movement. Controlling the effects of these activities depends on application of direct methods of slope stabilization or avoidance of areas of known instability.

Alpine Cirque Basins

These are high-elevation areas of low productivity due to harsh, cold climate with a short growing season and thin soils.

Snow Avalanche Paths

These are forested land areas that are considered not suited for management because of snow avalanche hazard. Below timberline, active areas are treeless strips, often following a gully. Less active areas may appear as strips of smaller trees, or strips of trees that are of a different species than those outside of the path. Runout zones may be outlined by changes in vegetation (Perla and Martinelli 1978).

"The damage caused by avalanches can be summarized as follows: The dislodgement of stones and soil; damage to pastures and forests, to buildings and communication routes, and finally danger to mankind and to animals." This sentence, written about a century ago by one of the pioneers in the subject matter, Coaz, is still valid (FAO 1985).

GIS PROCESS

Using a GIS for analysis of resource data to create management information is not new. Paper maps have always been available in some form. What is new is the computer power to consistently select a set of attributes and associated cartographic data from several different maps and process them as needed to create the desired interpretive map. This is where GIS analytical capabilities were used to prepare work maps for the Forest Plan.

Geographic data are organized into two generic classes consisting of cartography and attributes. These two data sets are related so there is a link between features represented by individual polygons, lines, or points and tables that quantify or describe the characteristics of the features.

Cartographic data consist of the location and topology of points, lines, and polygon features. Topology is the relationship among adjacent polygons, lines, and points in a cartographic data set.

Attribute data describe the characteristics of the cartographic data. For vegetation polygons, they describe the timber type and age class; for soil polygons, the relative stability hazard and control section soil texture.

To identify the location of land areas that are Tentatively Suited for timber management, an ARC/Info GIS was used to union (overlay) cartographic data for soils, landslides, slope, and vegetation. This created a data base with all possible polygon combinations that represent these data sets. The map representing Tentatively Suited lands was created by selecting from this master data base all polygons with the required attributes about soils, vegetation, and slope classes. The resulting maps showed locations and relative suitability for timber management.

MANAGEMENT ALTERNATIVES

Using the tentative land suitability base, a systematic, interdisciplinary approach involving resource specialists and public help was used to design and map management alternatives. An array of alternatives was developed to respond to public issues and forest problems and challenges through different management emphasis. Each alternative map shows a mix of Desired Future Conditions (DFC's). DFC's are mixes of compatible objectives and express the natural resource conditions that future site-specific field programs will be designed to achieve. The location and mix of DFC's for an alternative was based on the management emphasis and land characteristics.

SUMMARY

Forest Service management goals are to provide goods and services from the land without impairing land productivity or degrading water quality. Natural resource managers meet these goals using management elements that can be manipulated to achieve the corporate goals and objectives. These controllable elements are: job quality that is reflected in the way chosen management practices affect the ecosystem, quantity of inputs and outputs, location of the practices on the ground, and timing in implementing these practices. Weather, market economies, and societal preferences are uncontrollable elements. In forest planning, all of the elements must be considered, but only the controllable elements can be affected by management decisions.

Developing and communicating management goals and objectives through a planning process calls for aggregating basic resource data into management information. At the BTNF, lands were first classified as Tentatively Suited or Not Suited for forest management practices based on soil attributes, land slope, and vegetation. Lands were classified as Not Suited where technology is not available to prevent irreversible environmental damage from management practices. The remaining Tentatively Suited lands were categorized and management limitations noted. Additional resource data were used to develop standards and guidelines about quality, quantity, location, and timing.

Spatial data about the Tentative Suitability of individual resources were processed with GIS technology into maps for use in developing management alternatives. Final planning alternatives based on various sets of social values refined the management information about the controllable elements. An interdisciplinary process involving people from the Forest Service, other agencies, and interested citizens was used to design and plot management alternatives on the work maps. These alternatives were developed to show ways of responding to public issues and forest problems using a mix of DFC's.

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POTENTIAL USE OF GEOGRAPHIC INFORMATION SYSTEMS TO ENHANCE FOREST SOIL MANAGEMENT

Glen O. Klock

ABSTRACT

Computerized Geographic Information Systems (GIS) used as a resource management tool can provide opportunities for more effective input by soil scientists into the forest management decision process. Examples are given to show how GIS is now being used in forest soil management strategies. Future opportunities to use GIS models to predict the landscape position of current and potential forest productivity are discussed.

INTRODUCTION

The use of computerized Geographic Information Systems (GIS) has expanded into most natural resource management organizations in the past few years. Although GIS does not provide a new method, it does give faster and more visually effective opportunities for managers to display and analyze resource data. The use of GIS as a resource management tool can provide opportunities for more effective input into the forest management decision process by soil scientists.

Some opportunities where GIS outputs can be used to manage forest soils effectively include:

1. Create more spatially accurate and readable soil maps.
2. Provide a common medium to interact with other resource needs.
3. Analyze potential productivity of forest landscapes.
4. Analyze dynamics of forest productivity as affected by local management alternatives as well as other anthropogenic inputs.

Soils maps are a valuable tool for the analysis of the forest environment. However, too often the necessary soil map's scale is so large or the detail shown is so inadequate that the soil map is almost impossible to use. The storage requirements and cost to maintain needed field information on map sheets are often difficult to support. Once the proper soils data base has been installed in the computerized GIS system, maps and other ancillary data can quickly be displayed and plotted at the scale and level of detail needed. This capability gives the GIS an advantage over traditional methods.

Distortion of soil map data is another concern. It is disconcerting to assemble several soil map sheets together, overlay with a USGS quad sheet of the same scale, and find that the boundaries and stream features do not match. GIS provides the opportunity to correct this error and increases the credibility of the maps.

GIS is an effective tool to combine soil map data with other resource data to develop an area resource management plan or assessment. Three projects illustrate the cost effectiveness of the GIS system.

WALLOWA VALLEY PROJECT

In July 1989, 25,000 acres of the Wallowa-Whitman National Forest within the Wallowa Valley Ranger District were affected by wildfire. This fire, in the Sheep Creek drainage of the Upper Imnaha River, which drains into the Snake River, was either in an active timber sale or in a new sale area. To facilitate salvage logging, an environmental assessment was prepared. I was asked to provide soils information for that assessment.

I have had soils research projects near the affected area, but it appeared that an extensive field investigation would be required to evaluate the potential soil and water hazards that might be attributed to salvage logging. Because of the short timeline to accomplish the project, I chose GIS as a tool to support my evaluations.

A Soil Resource Inventory (SRI) was available for the area. As suggested earlier, the SRI map sheets were inadequate for use by themselves. Therefore, the SRI soil map sheets of the Canal Fire area were digitized and rectified to a UTM coordinate map base. In turn, the fire intensity and the hydrology maps of the area were digitized. Under the time frame, digital elevation maps (DEM) could not be secured, so the "slope breaks" by SRI unit were used to develop a slope map of the project area. In the same manner, surface erosion potential and mass soil stability potential maps were generated.

Once the GIS layers were established, a priority rating system was developed in which GIS modeling was used to project areas of soil management concern. For example, one objective was to map all areas with the common feature of high burn intensity, severe mass failure probability, and slopes over 35 percent. Likewise, common areas in which the erosion or mass failure potential were low were also identified. This computerized GIS method for the Canal project required less than 40 hours.

GIS maps showing areas of soil management concern were used as field guides to verify predicted hazards. Combining their use with aerial photos gave me an effective and efficient system to complete the project requirements.

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Because of the downstream fisheries, concern was expressed about the effects of the fire and potential salvage logging on water quality. Buffers of various widths based on stream class were suggested to protect the streams. The GIS helped the District staff determine how harvest volumes might be affected by various stream buffer widths. In addition, information on soil types sensitive to disturbance and potential erosion could be identified in the various proposed stream channel buffer zones.

GIS was also used to check if the salvage logging plans were consistent with the soil and water protection requirements in the environmental assessment. This "office evaluation" was quite effective, but difficult to complete as the logging engineers were not accustomed to using stable map bases to develop their logging plans.

DINKELMANN PROJECT

The second example of using GIS with soils data is the Seattle Water Department Dinkelmann Fire Salvage Project near Wenatchee, WA. Nearly 5,000 acres of Department forest land was affected by a 1988 fire. To assist in the salvage effort, low-cost information was needed on environmental concerns as well as appraisal information for both the timber sale offering and securing necessary Washington Forest Practices harvest permits. To meet this need, GIS soils maps were used to generate spatial statistics and maps for determining logging and transportation system design, defining environmentally sensitive areas, outlining erosion hazard areas, and developing regeneration plans. The use of GIS was again helpful in developing accurate and cost-effective land management data, mainly on soils information.

LAKEVIEW PROJECT

The third example focused on defining the affected environment in potential timber sales in the Lakeview District, Fremont National Forest, in Oregon. GIS was a useful tool to examine the soil types in each of the proposed cutting units. From the digital soils and hydrologic data, predictive soil and water effects information was generated for each proposed harvest unit. In addition, these data are being used in the KWCEA cumulative effects model (Klock 1985) to address the potential harvest impacts on water quality in one of the project watersheds.

Some of my earlier development work has shown that GIS can be used as a tool to map the landscape position of current and potential forest productivity. These maps were achieved by extending information gained by plot data across the landscape by the use of GIS modeling. While developing an analysis and consequent map of current productivity from plot data is rather obvious, developing some spatial distribution of potential site productivity by evaluating environmental factors becomes considerably more difficult.

EXTRINSIC-INTRINSIC FACTORS

Environmental factors that affect potential site productivity can be separated into two groups—extrinsic factors and intrinsic factors (Grier and others 1989). Extrinsic factors, such as climate, topography, and geology, provide the framework ecosystems develop on and are most influential in determining long-term regional productivity levels. Although all extrinsic factors can change or fluctuate over time, these factors are not influenced by the presence or absence of vegetation.

Intrinsic factors represent site characteristics that can affect productivity, influence ecosystem processes, and are at least semicontrollable by management. The most important intrinsic factors deal with soil properties because they can affect potential productivity. Soil-moisture-holding capacity, soil-nutrient status, and soil porosity or aeration are the factors that correlate best with site productivity (Carmean 1975; Ralston 1964).

It is generally not difficult to assemble in a GIS data that show the distribution of extrinsic factors across the landscape. Climate can be shown by precipitation distribution, degree days, etc. Topography can be easily defined by slope, aspect, elevation, and slope position. Geologic data can be defined to show soil parent material as well as the effects of soil-forming processes. Data of this type can be molded into a GIS spatial model to predict potential productivity, particularly of regional scale. This approach really does not address many of the specific intrinsic factors that affect site productivity nor those factors that may be controllable by management.

OPTIONS AVAILABLE

At this point there appear to be two options for the use of GIS to evaluate productivity at a site-specific level. The first method is to use the current vegetation community or plant association as an indicator of potential productivity. It is extremely difficult to map plant associations because of a host of factors such as fire and logging that may affect the current plant community condition. Spatially variable site characteristics collectively representing a set of environmental site conditions can, however, be determined for most plant associations. Modeling can be used, in turn, to project across the landscape an estimate of potential productivity. It is unlikely that enough soil intrinsic factor information will be available to project effectively how management activities will affect any given site's productivity by this method over a wide range of environmental conditions.

The second approach is through the use of research data to develop a scoring method of key spatial variable intrinsic soil properties—moisture-holding capacity, fertility, and aeration/drainage. From this information, using GIS, a productivity index projection can be made across the landscape. This approach, possibly supplemented with extrinsic site factor data, would be more effective in predicting the long-term effects of management activities than use of the first method using plant association indicators. This approach using GIS, however, is rather hypothetical at this time and further development work is needed.

Changes in climate and input of anthropogenic-produced pollutants have been suggested to affect forest productivity. The research community is currently attempting to link dynamic simulations of forest communities and Geographic Information Systems in an interactive environment to study these changes in our environment.

Undoubtedly these studies will require good spatial soils data. In turn, the products of this research can be expected to provide more opportunities to enhance forest soil management in the future.

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ESTIMATING FOREST PRODUCTIVITY FROM REMOTELY SENSED DATA AND TOPOGRAPHIC VARIABLES USING THE SPATIAL ANALYSIS CAPABILITIES OF A GIS

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ABSTRACT

Remotely sensed data (LANDSAT Thematic Mapper image having 7 bands) were geo-referenced, and radiometrically corrected. From 52 permanent growth plots located in the Lubrecht Experimental Forest, mean annual gross volume increment of bolewood in cubic meters/ha/year was functionally related to single bands, band ratios, and landscape variables (slope, aspect, and elevation). The terrain variables were obtained from a digital elevation model (DEM) using a Geographic Information System (PAMAP). The resultant regression equation was used to generate a thematic productivity map.

INTRODUCTION

To be able to predict the actual or potential productivity of forest stands measured in cubic meters/hectare/year is very useful to forest managers because they use productivity estimates to optimize harvest schedules, perform financial analyses, and select appropriate silvicultural practices. Not only do they want to know the amount but also the spatial distribution of this productivity. In addition they would like to obtain this information without the aid of extensive and expensive field work.

Because productivity varies spatially, such maps are very important documents. Digital productivity maps are even more useful because they link spatial with nonspatial data. These specific-purpose maps are often referred to as thematic maps, because they contain information about a single subject or theme and are drawn over a simplified topographic base (Burrough 1987).

Due to advances in satellite remote sensing technology in the last decade in combination with the availability of fast and relatively cheap microcomputing power, Geographic Information System (GIS) software packages have become available and affordable. These packages have the capability to store vast volumes of spatial data derived from a variety of sources, including satellite sensors that provide rapid and cheap data collection. With a GIS, managers can efficiently manipulate, spatially analyze, and most important, display these data according to user-defined specifications in the form of maps (Marble and Peuquet 1983).

The objectives of this study were to (a) develop a prediction equation that estimated actual annual aboveground bolewood production as a function of remote sensing and landscape variables, and (b) create a classified forest productivity map for Lubrecht Experimental Forest.

HARDWARE/SOFTWARE

All image data processing was performed on a Compaq 80386-20mHz microcomputer having a Cypher 1/2-inch magnetic tape drive, 20-inch RGB Mitsubishi monitor, and 660 MB CDC hard disk drive using ERDAS v7.3 software (ERDAS Inc. 1988). Spatial analyses and map production were performed on a Gateway 2000 80386-25mHz microcomputer having a 150 MB CDC hard disk drive and 14-inch NEC Multisync color monitor using PAMAP GIS v2.2 software (PAMAP Graphics Ltd. 1989).

STUDY AREA

Lubrecht Experimental Forest (LEF) is located in western Montana, approximately 56 km (35 miles) northeast of Missoula. In general it is a mountainous area, 10,900 ha (27,000 acres) in size, owned by the State of Montana and managed by the Montana Forest and Conservation Experiment Station of the University of Montana (Steele 1964).

Elevation ranges from 1,219 m at the western edge of Lubrecht to 1,700 m at the eastern edge of the forest (Teuber 1983).

Vegetation is considered to be typical of mid-elevation coniferous forests of west-central Montana. The forest area is well stocked with second-growth timber. Eight distinct timber types are present: Douglas-fir (*Pseudotsuga menziesii*), western larch (*Larix occidentalis*), ponderosa pine (*Pinus ponderosa*), lodgepole pine (*Pinus contorta*), Engelmann spruce (*Picea engelmanii*), subalpine fir (*Abies lasiocarpa*), cottonwood (*Populus trichocarpa*), and quaking aspen (*Populus tremuloides*) (Steele 1964).

The climate is characterized by wide fluctuations in temperature between winter and summer. Mean winter air

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temperature (December through February) is about 21 °F while mean summer air temperature (June through August) is about 59 °F. Annual precipitation is approximately 45.7 cm, with approximately 44 percent falling during the winter months (November through March) mostly as snow, and 24 percent falling during the summer (June through August) (Nimlos 1987).

The soil types of Lubrecht can be categorized into four very general groups (orders) (Nimlos 1987):

Order	Coverage
	Percent
Alfisols	9
Entisols	2
Inceptisols	86
Mollisols	3

GROUND DATA

Vegetation Information—One hundred and two permanent research plots distributed all over LEF, most of them established by former University of Montana Professor W. Pierce in 1960, provided the necessary vegetation data.

The size of the plots is 809.4 m^2 (0.2 acre). Measurements on all trees in each plot with a diameter at breast height (d.b.h.) larger than 11.43 cm (4.5 inches) (measured to the nearest tenth of an inch) have been taken every 5 years since 1960. In addition, tree height was measured to the nearest foot. These measurements were utilized to calculate the volume of each individual tree in each plot using a volume equation developed by Champion Timberlands, Rocky Mountain Operations. The form of the volume equation was:

$$V = 10^a * DBH^b * HT^c \tag{1}$$

where

V = volume of bole wood in ft^3
DBH = diameter at breast height in inches
HT = height in feet
a, b, c are constants depending on tree species and age.

The tree volumes were indirectly estimated from d.b.h. and height measurements, converted to cubic meters, and summed for each plot by species for each of two measurement years, 1970 and 1980. Finally, the volume difference (1980 volume minus 1970 volume) was divided by 10 and then converted to a per-hectare basis. Annual volume estimates ranged from 0.614 to 6.551 m^3/ha/year with a mean of 3.171 m^3/ha/year.

Landscape Information—A series of panchromatic (black and white) aerial photos at 1:24,000 scale, taken on August 23, 1952, containing the pinholed research plot locations, section lines, and section corners were georeferenced using the PAMAP GIS. Through the spatial analysis capabilities of this GIS the independent variables of interest (slope, aspect, elevation, and soil type) were extracted for all plots from a digital elevation model (DEM).

REMOTELY SENSED DATA

Satellite remotely sensed data were obtained from a LANDSAT 5 overpass on July 18, 1984. They were available from another study (Nemani 1987). The existing scene is a seven-band LANDSAT Thematic Mapper (TM) image with 30-by-30-m spatial resolution in all bands except band 6 which has 120-by-120-m. The characteristics of these bands are listed in table 1 (Lillesand and Kiefer 1987).

GEOMETRIC CORRECTION

Before a raw digital image can be used as a map, signifcant geometric distortions (due to the Earth, the satellite, the orbit, and the image projection) must be removed. Using the ERDAS software the LANDSAT image was registered in the Universal Transverse Mercator (UTM) coordinate system using six distinct, clearly visible ground control points. At the same time the image was geometrically corrected using the nearest neighbor resampling technique. This method offered the advantage of computational simplicity and did not alter the original input pixel values (Lillesand and Kiefer 1987). Upon subsequent testing a registration accuracy of one pixel was achieved. This is the optimum expected with manual registration for carefully selected control points (Schowengerdt 1983).

RADIOMETRIC CORRECTION

Correction for Atmospheric Scattering (Haze Removal)—TM band 7 is essentially free from atmospheric effects. Utilizing a small body of homogeneous, deep, nonturbid water (lake in Jones meadow), the histogram of its reflectivity values in band 7 was examined. The minimum reflectance value of band 7 was found to be zero. Then the minimum reflectance values in bands 1, 2, and 3 of the lake were used as offsets to adjust the reflectance values of all pixels in bands 1, 2, and 3, respectively.

Correction for Topographic Effects—Variations in surface slope and aspect cause variation in spectral radiance, which is most pronounced in areas of rugged terrain

Table 1—LANDSAT Thematic Mapper spectral band characteristics

Band	Wavelength	Spectrum
	μm	
1	0.45 - 0.52	Blue-green
2	.52 - .60	Green
3	.63 - .69	Red
4	.76 - .90	Near-IR
5	1.55 - 1.75	IR
6	10.4 - 12.5	Thermal
7	2.08 - 2.35	Far-IR

(Holben and Justice 1981). Band ratioing is frequently used to remove undesirable effects. In this study the irradiance of any slope as well as that of flat ground was calculated, and a ratio of these two radiation values was used to normalize the reflectivity values for all bands.

Since the ground plot size was 809.4 m^2, slightly smaller than the TM pixel size (30- by 30-m = 900 m^2), a nine-pixel area (a three-by-three block) was manually identified to represent each plot in order to compensate for registration errors and also to take into account cases where the plot was located on two or more pixels. Mean corrected reflectance values for each three-by-three block were extracted for all seven TM bands using the ERDAS software.

ANALYSIS

From the original 102 established plots at LEF, only 52 were used for the development of the productivity model. Sixty plots were excluded for various reasons such as questionable location, missing or erroneous tree measurements, site disturbance, slope/aspect discrepancies, and biased reflectance values strongly influenced by nearby roads, water, or meadows.

The initial data set contained radiometrically corrected reflectance values for all seven TM bands and the landscape variables slope, aspect, and elevation. These later variables are known to affect local air and soil temperature, moisture, and soil drainage (Spurr and Barnes 1980). Various studies have shown that productivity estimation can be improved considerably when these predictor variables are included (Cook and others 1987a,b; Fox 1985; Tom and Miller 1979). A number of interaction terms between slope and aspect were created to capture the effect of insolation. Site quality was captured by obtaining a number of soil variables through the PAMAP GIS from database records linked to soil series polygons. Most of them were simply descriptive variables such as soil series, parent material, texture, fragment description, and common habitat types associated with each soil series. However, one variable, Douglas-fir site index, could be used as a proxy for soil productivity.

Next, a number of TM band ratios were constructed. Band ratios not only minimize differences in illumination due to topography, but also emphasize differences in the slopes of the spectral reflectance curves between the bands forming the ratio (Lillesand and Kiefer 1987; Sabins 1987). These ratios uncover physical or chemical characteristics of the sensed area not distinguishable by the use of any single band. For example, a band4/band3 (B43) ratioed image has proved very useful in many studies in discriminating between areas of healthy (high values) and stressed (low values) vegetation.

Deciding which two bands to ratio is not always a simple task (Jensen 1986). Either a subjective or objective approach is used. In this case an objective method based on the concept of an optimum index factor (OIF) developed by Chavez and others (1982, 1984) was tried first and then discarded in favor of a subjective approach.

In addition to simple band ratios, more complicated ratios called vegetation indices, were also computed. The rationale behind vegetation indices is to reduce the seven TM band values to a single number per pixel, for qualitatively and quantitatively assessing characteristics such as leaf area, biomass, or productivity (Jensen 1986; Perry and Lautenschlager 1984). The vegetation indices initially constructed were of the normalized difference type:

$$NVIxy = (BxNORM - ByNORM)/(BxNORM + ByNORM)$$

where

$BxNORM$ and $ByNORM$ = normalized bands x and y.

Only NVI31, NVI32 and NVI43 were calculated.

Stepwise multiple regression analysis was employed to produce a reliable productivity prediction equation. The PAMAP GIS utilized this prediction equation on a pixel-by-pixel basis to produce a classified productivity map.

RESULTS AND DISCUSSION

Simple linear correlations between mean actual productivity and single-band corrected reflectance values were all negative. Excluding band 6, band 2 had the highest correlation ($r = -0.306$) and band 4 had the lowest ($r = -0.206$). Theory suggests that TM band 4 radiance (near-infrared) should be positively related to coniferous forest biomass and hence productivity. An investigation revealed that most plots had open canopies with an understory of broadleaf shrubs or grasses and litter in small quantities. These added noise to the reflectance values. When these plots were excluded, the relationship became positive.

These modest correlations were expected because satellite sensors sense green biomass, not dead bolewood. Many studies have shown that there is a relationship between green biomass and the spectral data received by the sensors (Cook and others 1987; Peterson and others 1987; Plummer 1988; Spanner and others 1990). Further, it has been shown that green biomass depends loosely on the amount of standing tree woody biomass. Productivity is a complex phenomenon. Two forest stands that seemingly have the same green biomass may have different productivities, depending on their age, stand structure, site quality, and understory composition.

The "best" productivity model was:

$$\begin{aligned} PROD = \underset{(1.916)}{11.363} - \underset{(0.00150)}{0.00643} * ELEV - \underset{(0.157)}{0.539} * B1NORM \\ + \underset{(0.219)}{0.437} * B2NORM + \underset{(3.533)}{14.809} * NVI32 \quad (2) \end{aligned}$$

where

PROD = predicted productivity (in m^3/ha/yr)
ELEV = elevation (in m)
B1NORM = normalized band 1
B2NORM = normalized band 2
NVI32 = normalized difference vegetation index

with $n = 52$, adj $R^2 = 0.410$, SEE = 1.116, % SEE = 35.2.

An analysis of the residuals revealed that no underlying assumptions had been violated.

The selection of ELEV as an important predictor variable was expected. Its negative relationship with productivity was theoretically justified, because at the latitude of the study area elevation affects significantly the number of

frost-free days, thus limiting the growing season. On the other hand, slope and aspect did not correlate with productivity, although it is known that an aspect-slope combination is a critical factor in plant growth. This anomaly may be due to the fact that these plots were located so as to have relatively high productivity in spite of their aspect, to have relatively gentle slopes, and to be in easily accessible areas next to roads or trails, since they were to be remeasured every 5 years. The lack of representation of a wide range of slopes and mainly a lack of plots with a variety of productivity levels in different slope-aspect combinations resulted in very poor correlation between productivity and slope-aspect.

TM bands 1, 2, and 3, which contribute to the prediction of productivity, all sense in the visible part of the electromagnetic spectrum. According to Lillesand and Kiefer (1987) band 1, which is designed primarily for water body penetration, is also useful for soil/vegetation discrimination, forest type mapping, and cultural feature identification. Band 2 is designed to measure the green reflectance peak of vegetation and is useful for vegetation discrimination and vigor assessment. Band 3 is designed to sense in a chlorophyll absorption region, aiding in plant species identification. The selection of these bands is in agreement with their stated design objectives. Soil generally has higher reflectivity in bands 1, 2, and 3 than vegetation. Thus plots with high vegetation cover would be expected to have lower band 1, 2, and 3 values. Assuming that more vegetation would mean more productivity, the negative coefficient for band 1 agrees with this fact. After the inclusion of band 1, band 2 enters positively even though it was correlated negatively with productivity. Finally, the normalized difference index NVI32 had a good positive correlation with productivity. This index is based on two bands that are specifically sensitive to chlorophyll content and as such its inclusion in the prediction equation would be expected to help quantify vegetation vigor in the plots.

Soils, as characterized by DF site index, proved to have no correlation (adjusted R^2=0.00) with productivity. This may be due to a lack of information regarding soil depth, which is a critical factor for root development and water holding capacity, especially in a water-limited environment (Nimlos 1989).

The derived productivity equation leaves a significant amount of variation unexplained. However, its adjusted $R^2 = 0.410$ is similar in magnitude to the values achieved by Cook and others (1987a,b) ($R^2 = 0.39$ to 0.42) in their studies on hardwood vegetation in Illinois and New York. It should be noted that the horizontal or vertical arrangement of the leaves in the canopy determines the direction of the reflectivity (Plummer 1988). Conifers are mainly erectophile vegetation, so a bigger amount of radiation is scattered by their canopy. More scattering means less direct reflectivity, so for conifers there is a weaker relationship between the amount of vegetation and reflectance values than for planophile vegetation.

Some of the unexplained variation may be due to inaccuracies in the data. Considerable thought was given to the identification of probable sources of such problems. They generally fall in two categories. First, there are inaccuracies due to probable human errors in the actual measurements of tree d.b.h. and especially height in Pierce's plots. Also, volume estimation errors (due to possible inaccuracies in the volume equations) may be included in this category.

The second category of inaccuracies results from the very fine 30-m resolution used for this study. The lack of perfect one-to-one correspondence between image pixels and plots on the ground (parts of a plot can lie in up to four pixels) necessitated the use of a value for each band that was the average of the values of the nine closest pixels to the plot's position. Any variations in vegetation, or gaps in the areas next to the plot, could add noise to the data. Small possible errors in plot positioning (in spite of the care that was exercised) could further contribute to the inclusion of erroneous information in the data set. If only band values from the central of the nine pixels had been used, the effect of even the smallest plot positioning error would be much greater.

Gaps in the overstory vegetation can have a significant confounding effect on the information that can possibly be extracted from the satellite data. Canopy gaps can allow the sensor to sense bare soil, rock outcrops, grasses, needles, or broadleaf understory. The effects of this understory on the image can be very significant (Spanner and others 1990). If understory information was available it would permit improved productivity estimation. However, such information is not readily available for the whole forest and its use would defeat the objective of productivity mapping without visiting the site.

The image acquisition date (July 18, 1984) was in the middle of a hot, dry summer and thus trees were already under water stress. TM bands 5 and 7 have proven to have a strong negative correlation with the amount of moisture in green vegetation by other studies. If an image from the beginning of the growing season had been used, there would be additional information about vegetation moisture that might help in the prediction of biomass and productivity.

CONCLUSIONS

The predictor variables associated with the final productivity equation accounted for 41 percent of the variation in Y. In judging its predictive capabilities, one should take into consideration the accuracy that alternative productivity prediction methods can achieve, as well as the practicality of applying such methods over large areas.

A number of weaknesses were identified that were probably responsible for the amount of unexplained variation. The major weakness in the data set was lack of representation of the whole range of encountered productivity, especially in relation to various steep slope-aspect combinations. Future application of the methodology used in this study would greatly benefit if the data set for the creation of the predictive equation included data points of this type.

Additional suggestions for future studies include selection of plots away from roads, rivers, or water bodies that can affect the reflectivity values recorded by the sensors, and careful selection of the image acquisition date so that it will include as much of the desirable information as possible. If plots are to be established for productivity measurements they should be identified from the image in a way that they will represent as much of the observed image

variation as possible. Location of each plot should be selected within a homogeneous area in the image so that the pixels representing it will be surrounded by pixels of nearly identical values, removing any possible sources of noise.

Finally, future applications should be expected to benefit greatly from advances and cost reductions in Global Positioning Systems (GPS), which at this time appear to be coming of age. Positioning of plots could become extremely accurate, which is a necessary condition for taking full advantage of the high resolutions offered by current and future satellite sensors.

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RESPONSES AND FEEDBACKS OF GLOBAL FORESTS TO CLIMATE CHANGE

Robert K. Dixon

ABSTRACT

The accumulation of greenhouse gases in the atmosphere over the past century is projected to cause a warming of the Earth. Climate change predictions vary by region and terrestrial biosphere response and feedbacks will be ecosystem specific. Forests play a major role in the Earth's carbon cycle through assimilation of CO_2, storage of carbon, and emission of greenhouse gases. Simulation models have been employed to examine the possible responses to climate change of global forest ecosystems. Major shifts in forest species distribution and composition are predicted in response to projected climate change within the next 50-80 years. The range of some species is expected to shift dramatically in biomes worldwide. Savanna-type vegetation could replace some forests under the more extreme climate change predictions in temperate latitudes. The ultimate response and feedbacks of forests will be influenced by the direction and magnitude of climate change, site quality, and other stress agents. Establishment of new forests and implementation of management practices could potentially be used to sequester significant amounts of atmospheric CO_2. Preliminary evidence suggests the terrestrial biosphere could be managed to reduce accumulation of greenhouse gases in the atmosphere and mitigate negative impacts of climate change.

INTRODUCTION

Greenhouse gases (for example, CO_2, CH_4) produced from anthropogenic and biogenic emissions are accumulating in the atmosphere. The concentration of atmospheric CO_2 is projected to double from preindustrial levels by late in the 21st century. Infrared radiation trapped by greenhouse gases in the troposphere is expected to influence global climate. General Circulation Models (GCM's) of climate change project the average temperature of the Earth's surface will increase 1.5 to 4.5 °C and influence regional frequency and distribution of precipitation (Schneider 1989a,b). Considerable uncertainty exists regarding the magnitude of global climate change, particularly projections of regional responses and feedbacks by terrestrial ecosystems, especially forests (Dickinson 1989; Sedjo and Solomon 1989).

THE GLOBAL CARBON CYCLE

The accumulation of CO_2 in the atmosphere in recent decades has increased interest in the global carbon cycle (Tans and others 1990). The net increase in atmospheric CO_2 is the result of greater carbon release than that being removed by the terrestrial biosphere and marine systems (fig. 1). Two sources of CO_2 are especially significant, the combustion of fossil fuels and global deforestation. The flux of carbon through the terrestrial biosphere (for example, by plant photosynthesis, respiration, and decomposition) is approximately 100 Gt annually. Oceans are large pools of global carbon, but annual net flux with the atmosphere is relatively low.

FORESTS AND THE GLOBAL CARBON CYCLE

Terrestrial ecosystems, especially forests, play a major role in the Earth's carbon cycle through assimilation of CO_2, storage of carbon, and emission of carbon gases to the atmosphere (fig. 1; table 1). Forests have high rates of ecosystem productivity (the amount of carbon photosynthesized less that respired) compared to most other ecosystems. The world's forests hold approximately 90 percent (about 740 Gt) of all aboveground terrestrial carbon, and 40 percent (about 570 Gt) of all belowground terrestrial carbon (Waring and Schlesinger 1985).

The cumulative global net release of carbon to the atmosphere due to forest clearing, from 1860 to 1980, is estimated to range from 135 to 228 Gt (Woodwell and others 1983). Between 1.8 and 4.7 Gt of carbon were released from biotic sources in 1980 alone, of which 80 percent was due to deforestation (Detwiler and Hall 1988). Carbon release from forest burning in 1980 has been estimated to be 50 percent of the annual atmospheric increase of 3 Gt. The clearing of forest land for agriculture, especially within the tropical latitudes, is now the largest source of carbon released to the atmosphere from the biota and soils globally. However, actual estimates differ by a factor of 2-3, due to differences in estimated rates of clearing tropical forests (Detwiler and Hall 1988; Woodwell and others 1983).

If deforestation increases in proportion to population growth, the biotic release of carbon will reach about 9 Gt/yr before forests are exhausted in the next century (two times the current fossil fuel emissions) (Postel and Heise 1988). Detwiler and Hall (1988) imply that of the 5.1-7.5 Gt

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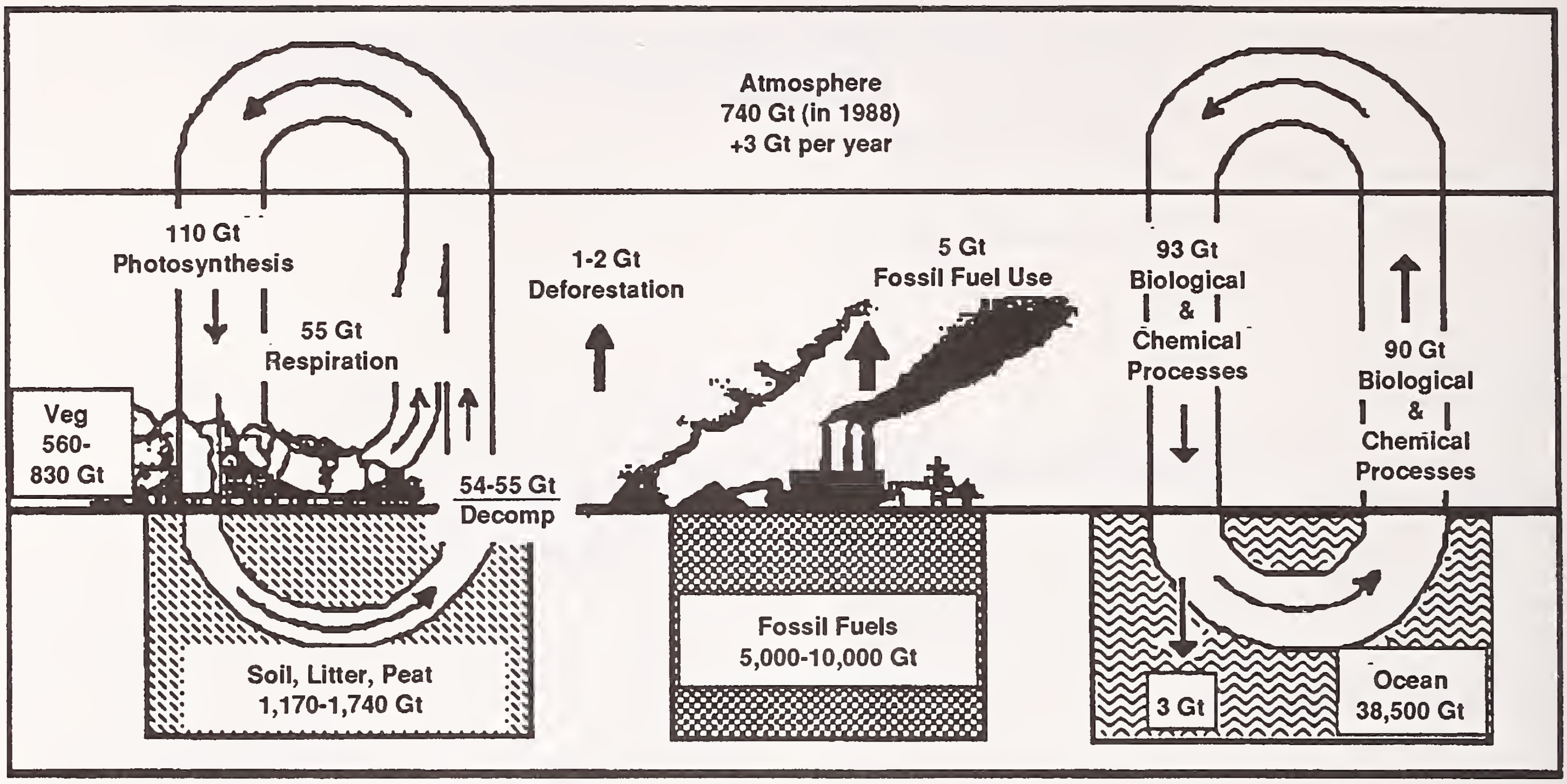

Figure 1—The global carbon cycle including pool size and flux for terrestrial and marine systems (adapted from Schneider 1989a).

Table 1—Anthropogenic and biogenic emissions (1980 estimates) of carbon to the atmosphere and potential sequestration of carbon dioxide (CO_2) by establishment and intensive management of world forests (Schroeder and Ladd 1990; Tans and others 1990; Woodwell and others 1983)

	Range of estimates
	Gt/yr
Carbon source	
Fossil fuel	4.8 - 6.6
Deforestation and other biotic sources	1.8 - 4.7
Carbon sink	
Forest establishment-500 Mha	3.5 - 4.0
Forest management-300 Mha	0.5 - 1.5

released annually by fossil fuel and deforestation, somewhere between 0.3 to 2.8 Gt is assimilated in the terrestrial biosphere. Variation in the annual atmospheric concentration of CO_2 is greater in the northern hemisphere in part due to a greater land mass and vegetation (forest) cover (Tans and others 1990). Thus, the terrestrial biosphere plays a major role in the global carbon cycle.

FOREST RESPONSE TO GLOBAL CHANGE

Large uncertainties exist regarding forest response to climate change (Jarvis and others 1989; Sedjo and Solomon 1989). As atmospheric concentration of CO_2 increases and climate change events unfold, forests could become either a net source, or sink, of carbon. Climate change will almost certainly cause some forest species in selected regions to decline and migrate (Urban and Shugart 1989; Woodman and Furiness 1989). Ecosystem shifts in species distribution and composition are projected to occur within the next 50-80 years. For example, the range of loblolly pine in the southern United States is predicted to shift north several hundred miles. A long-term decline in productivity of some forest types could occur and timber production, biotic habitat, yield of water, site quality, and recreation opportunities may be altered.

Based on GCM estimates of climate change associated with a doubling of atmospheric CO_2 by the year 2050, and the subsequent redistribution of vegetation, the world's forests could experience a substantial change in distribution and composition (an areal increase or decrease). Estimates of global forest redistribution vary widely between GCM's used to estimate climate changes (10 percent decrease in areal coverage with the GFDL model, 60 percent increase with the GISS model) (Emanuel and others 1985; Prentice and Fung 1990). Consequently, the resulting biosphere

feedbacks to climate could be negative or positive. The major negative feedback to climate change is CO_2 enrichment of vegetation, while positive feedbacks include biogenic emissions of greenhouse gases (for example, CH_4, NMHC, H_2O). Boreal, temperate, and tropical forests will respond to climate change differently, and must be managed differently to adapt to a changing environment (Smith and Tirpak 1989).

Large areas of world forests (especially temperate and boreal) could experience water stress due to warming and drying, which could lead to widespread forest decline thus producing a potential major source of atmospheric CO_2. Timing of changes in forest condition is unclear, but based on past climate-related events (as with extended droughts) alterations could be manifested in the first half of the next century. Alternatively, atmospheric CO_2 enrichment could increase forest productivity and water-use efficiency (Mooney and others 1990). These positive ecophysiological effects could help forests adapt to global warming. The ecophysiological response of broadleaf and conifer seedlings to CO_2 enrichment is different in short-term studies (Mooney and others 1990). Differences in seedling biomass, leaf area, root-shoot ratios, water-use efficiency, and nutrient-use efficiency have been ascribed to short-term CO_2 enrichment. Long-term responses to CO_2 exposure are largely unknown. Limitations of nutrient and water resources may not preclude plant growth responses to CO_2 enrichment (Norby and O'Neill 1989).

MANAGEMENT OF FORESTS TO MITIGATE GLOBAL CLIMATE CHANGE

Forest ecosystems can be managed to increase CO_2 assimilation via photosynthesis and temporarily store large amounts of carbon (table 1; Schroeder and Ladd 1990). Establishment of 500 million ha (area approximately the size of Australia) of new forests worldwide could fix 2.5 Gt of carbon/yr aboveground and another 1-1.5 Gt/yr belowground (Trexler 1990; Wood and others 1984). Estimates of aboveground carbon assimilation rates range from about 1 Gt/yr in boreal forests to about 8 Gt/yr in tropical forests. It would take about 25 years to plant 500 million ha assuming 20 million ha/yr could be planted. Of course, many assumptions are included in these estimates, such as: (1) no large-scale forest decline due to global change, (2) level population growth and no land-use changes, (3) reforestation is not offset by deforestation, and (4) effects of CO_2 enrichment are negligible.

Intensifying silvicultural practices on existing land in boreal, temperate, and tropical forests could result in an additional 0.5 to 1.5 Gt/ha/yr of carbon being fixed (Schroeder and Ladd 1990). The area of forest land where silviculture could be intensified is about 300 Mha, globally (10 percent of the world's closed forests) (Wood and others 1984). Thus, the carbon-sequestering rate could be stimulated by over 10 percent. Logging debris and soil organic matter could also be managed to maintain or sequester significant amounts of carbon.

Given the summary of estimated global carbon sources and sinks shown in table 1, the prospects for reducing the accumulation of greenhouse gases (for example, CO_2) in the atmosphere through forest management appear promising. However, many biologic, socio-economic, and political barriers exist and global management of the carbon cycle is probably decades in the future. Anthropogenic and biogenic emissions are predicted to increase dramatically in the next century. A combination of efforts to slow accumulation of greenhouse gases in the atmosphere including a reduction in fossil fuel combustion, slowing deforestation, and establishment of new forests may be practical and complementary alternatives for developed and developing nations.

CONCLUSIONS

Forests play a central role in the global carbon cycle. Although estimates of global carbon pool size and flux vary, forest distribution and productivity influence concentration of greenhouse gases (for example, CO_2, CH_4) in the atmosphere. Variation in annual atmospheric concentration of CO_2 is greater in the northern latitudes in part due to a greater land mass and vegetation (forest) cover. Substantial scientific uncertainty exists regarding the role of forests in global change and the carbon cycle. Preliminary evidence suggests forests could be managed worldwide to reduce the accumulation of greenhouse gases in the atmosphere and possibly mitigate the negative impacts of global change. Considerable research is required to reduce the large uncertainties regarding the global carbon cycle and projected climate change events.

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Speakers answered questions from the audience after their presentations. Following are the questions and answers on this topic:

Q.—Will fossil fuel burning continue beyond the year 2010 (or will the supply of fossil fuel be totally consumed)?

A.—Global fossil fuel reserves are predicted to be available beyond year 2010. Large reserves are available on several continents, especially Asia.

Q.—A recent report from NASA stated that no measurable atmospheric temperature change was observed by satellite in the last 10 years. Is the Earth warming?

A.—Short-term measurements (for example, 10 years) of climate patterns are highly variable and can be misleading. The long-term atmospheric temperature trends (previous 120 years) reveal the Earth has warmed 0.5-0.8 °C according to the Intergovernmental Panel on Climate Change (IPCC).

Q.—Is the increase in atmospheric CO_2 the sole responsibility of man?

A.—No, biogenic and anthropogenic sources of gases both contribute to atmospheric chemistry.

Q.—Could global climate change be attributable to natural phenomena (for example, volcanism) rather than the influence of man?

A.—Major catastrophic events in the terrestrial biosphere significantly contribute to changes in atmospheric chemistry. The source-sink relationships of the terrestrial biosphere remain a significant research question.

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PREDICTING FOREST CHANGES ASSOCIATED WITH CLIMATE WARMING: POTENTIAL USES OF GIS TECHNOLOGY

David L. Verbyla

ABSTRACT

Research is needed to understand potential local forest responses to climate warming. Geographic Information Systems (GIS) technology can be used to establish permanent plots at hot or cold extremes of vegetation zones—areas where immediate effects of climate warming may be detectable. Empirical studies using GIS technology and solar radiation indices can be used to test hypotheses of expected forest changes associated with recent climate warming. However, because ambient carbon dioxide is likely to increase and may ameliorate the effects of climate warming on forests, long-term predictions from climate warming studies should be interpreted with caution.

INTRODUCTION

Most research that predicts forest changes associated with climate warming has been regional or global studies using computer simulation models (for example, Davis and Botkin 1985; Dale and Franklin 1989; Pastor and Post 1988). However, much of our forest management occurs at the forest or ranger district level. How will climate warming affect soil and vegetation processes at this scale? For example, will Armillaria root disease within a Douglas-fir habitat type become a more serious problem if the climate warms and drought stress in Douglas-fir increases? Will mountain pine beetle in lodgepole pine decline because of a less favorable (hotter) microclimate, or will lodgepole pine trees become more stressed and therefore more susceptible to beetle attacks? Will nitrification rates significantly increase in spruce-fir habitat types if the climate warms? The objective of this paper is to outline an empirical approach aimed at addressing such research questions.

POTENTIAL CHANGES IN VEGETATION DISTRIBUTION

In the past, forest communities have shifted in response to climate warming during the Pleistocene and Holocene periods (Baker 1983; Van Devender and Spaulding 1979). With CO_2-induced climate warming, similar vegetation zone migrations in latitude and elevation are expected (Emmanuel and others 1985; Payette and Filion 1985).

Where should permanent plots be established to monitor these shifts in vegetation zones? If permanent plots are randomly established, and are therefore mostly on "normal" sites (near the center of a species' distribution), the effect of climate warming might not be detectable for decades or centuries.

However, permanent plots established at the hot or cold extremes of a vegetation zone would be more sensitive to immediate climate warming. For example, at its northern distribution limit, white spruce (*Picea glauca*) has expanded significantly in response to recent climate warming (Payette and Filion 1985). To monitor the early effects of climate warming, a method is needed to efficiently delineate the "hot" and "cold" extremes of vegetation zones.

PREDICTING CHANGES ASSOCIATED WITH CLIMATE WARMING

How will forest insect and disease problems change with climate warming? How will soil properties and processes change? These types of questions can be addressed by examining current conditions within stands sampled from a range of "cool" to "hot" sites.

For example, Amman and others (1988) hypothesized that microclimate is a principle factor controlling mountain pine beetle (*Dendroctonus ponderosae*) infestations in lodgepole pine (*Pinus contorta*). Will mountain pine beetle infestations in lodgepole pine become less severe if the climate becomes hotter? Geographic Information Systems (GIS) can be used to efficiently select sample stands from "cool," "normal," and "hot" sites within the lodgepole pine series. Trends in mountain pine beetle density could then be examined across the continuum of "cool" to "hot" lodgepole stands and predictions could be made about future beetle infestations in response to climate warming.

How can GIS be used to rank stands as "cool," "normal," or "hot" sites? Digital Elevation Models (DEM) can be purchased from the U.S. Geological Survey for certain areas of the Rocky Mountains. If DEM data are not available, they can be derived by digitizing contours from topographic maps. They also can be generated photogrammetrically with an analytical stereoplotter. Given a DEM, the

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potential solar radiation of any site can be computed as a function of latitude, slope gradient, slope azimuth (aspect) and Julian day (Flint and Childs 1987; Garnier and Ohmura 1968; Harrington 1984; Kaufmann and Weatherred 1982; Lee and Baumgartner 1966; Swift 1976). Each stand can then be assigned a "radiation index" value by integrating potential solar radiation over the water-limited season. Stands can be selected while controlling for other important factors, for example, parent material, age class, and species composition. Therefore, changes in radiation indices among stands are assumed to be the dominant factor that influences the microclimate of each stand. This is similar to comparing stands from north-facing ("cool") and south-facing ("hot") slopes where each stand is similar in terms of parent material, age class, and species composition.

Potential solar radiation has been used successfully in many diverse areas including prediction of the distribution of frozen soils (Zuzel and others 1986), prediction of rock glacier development (Hassinger and Mayewski 1983), prediction of watershed runoff yield (Lee 1964), and vegetation ordination studies (Dargie 1984; Parker 1989).

The GIS approach has several advantages. Stands can be efficiently selected to control for other confounding factors. For example, it is easy with a GIS to select all stands from a Douglas-fir series, on limestone parent material, with a certain basal area and age class. Second, this approach of using a radiation index to rank stands is an empirical approach that can be used to test hypothesis generated by theoretical computer simulation models. Third, this empirical approach can be used to address research questions that would take decades to answer with controlled experimentation.

POTENTIAL PROBLEMS

Many technical problems need to be resolved. Potential solar radiation varies daily. What is the appropriate period to integrate radiation over—the entire growing season, the period of maximum drought, or the entire year? East- and west-facing slopes receive the same amount of solar radiation; the only difference is that the west-facing slopes receive most of the radiation loading after solar noon when plants are often under drought stress. This timing factor must be incorporated in a useful solar radiation index. Elevation must also be factored out; a south-facing slope at 9,000 feet might be "cooler" than a north-facing slope at 7,000 feet.

Another problem is that long-term climate warming will be associated with an increase in ambient carbon dioxide. Because of this increased carbon dioxide, all models that predict changes in vegetation and soil factors but ignore the effects of increased CO_2 are speculative. For example, a computer simulation analysis conducted by Revelle and Waggoner (1983) suggested that watersheds in the western United States will suffer 40 to 75 percent **reduction** in streamflow due to climate warming. However, Idso and Brazel (1984), using the same computer model (but incorporating antitranspirant effect of CO_2), estimated that there would be an **increase** of 40 to 60 percent in streamflow.

The effect of increased CO_2 on ecosystem changes is poorly understood and yet is important in predicting forest and soil changes that may occur due to climate warming. For example, tree growth may actually increase at some sites due to the fertilization effect of increased carbon dioxide. The growth rates of subalpine trees in Nevada and California have been reported that exceed growth rates expected due to climatic trends but is consistent with increased carbon dioxide concentrations (LaMarche and others 1984).

Increased carbon dioxide may also ameliorate the effects of water stress in some plant species. For example, Tolley and Strain (1984) found that sweetgum (*Liquidambar styraciflua*) seedlings exposed to water stress and grown at elevated CO_2 conditions had final dry weights significantly greater than seedlings grown under well-watered and normal CO_2 conditions. Hurt and Wright (1976) found similar results with knobcone pine (*Pinus attenuata*) and Coulter pine (*Pinus coulteri*).

Despite these problems, research is desperately needed for rational forest management in the event of climate warming. Until now, much of the climate warming research has been on a global and regional level. We need to begin research on the local forest level. GIS technology and radiation indices are tools that can be used at this local level address the question: what trends can we expect in our forests if the climate becomes significantly hotter in the 1990's?

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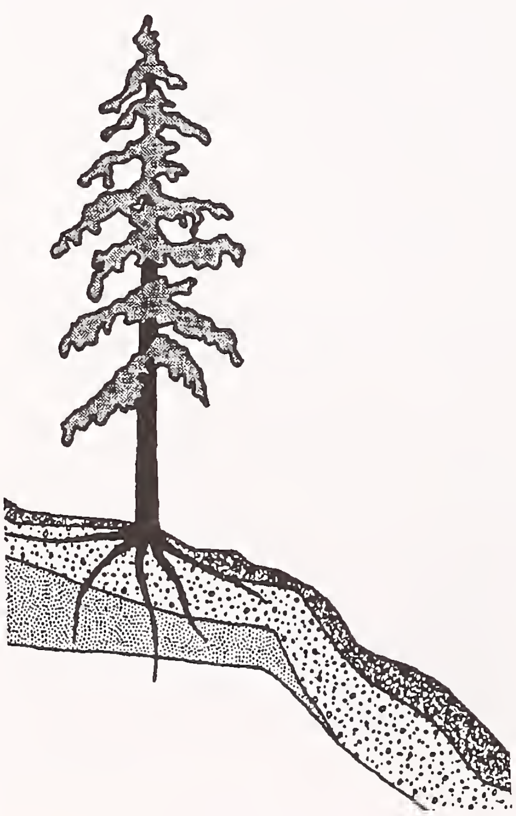

COMPARISONS OF TREE HEIGHT GROWTH ON BROADCAST-BURNED, BULLDOZER-PILED, AND NON-PREPARED SITES 15 TO 25 YEARS AFTER CLEARCUT LOGGING

Bob Bosworth
Dan Studer

ABSTRACT

Tree height growth was compared on clearcuts that were broadcast burned, bulldozer piled, or had no preparation for regeneration in the Bonners Ferry Ranger District, northern Idaho. Differences in heights and growth are related to treatment. Height growth on burned sites exceeded that on piled sites in 8 of 9 years. Questions arise as to effects on site productivity of piling. Reasons for differences include nutrients available to regeneration related to site preparation method, and compaction effects from piling.

INTRODUCTION

The Bonners Ferry Ranger District of the Idaho Panhandle National Forests lies in northern Idaho. Generally, it is bounded by Canada to the north, Montana to the east, the crest of the Selkirk Mountains to the west, and extends south to the Boundary/Bonner County line. Elevations range from 2,500 to over 6,000 feet. Habitat types range from those of the relatively warm, dry Douglas-fir (*Pseudotsuga menziesii*) series to those of the cool, moist subalpine fir (*Abies lasiocarpa*) series (Daubenmire and Daubenmire 1968).

Subsoils in the district are dominated by glacial tills from both continental and alpine glacial action. Some are subsoils formed from glacial-fluvial outwashes. Most of the surface soils are volcanic ash in origin. This ash cap is a very good medium for growing trees (Ford 1985).

Clearcutting as a regeneration system has been practiced since as early as 1916 (Roe and DeJarnette 1965) in the district. However, use of the clearcut system became increasingly routine from the late 1950's to the mid 1970's. Most of this cutting has been in overmature to decadent timber stands where large quantities of slash and debris dictate some sort of site preparation to achieve adequate regeneration and to meet fire hazard reduction objectives. Broadcast burning and bulldozer piling have been the most frequent treatments. Some stands have had no site preparation treatments.

The district is continually juggling its program, considering the limited times suitable for broadcast burning and the total acres to be treated for planting each year. Typically, debris on slopes less than 35 percent is piled, while steeper slopes are burned. In years of larger programs, bulldozers are moved onto even steeper slopes to balance the program and get all sites prepared in a timely manner. With increasing concerns over air quality, there is more pressure to move away from broadcast burning and to piling and burning to reduce impacts on airsheds.

As our knowledge base on the complex interactions of our forest ecosystems expands, the questions of short- and long-term impacts on growth related to site treatments are repeatedly asked. Bonners Ferry District personnel have been intensively monitoring progress of regenerating stands for many years. We wondered if the data from monitoring might be able to shed some light on the effect of treatment on regeneration performance. We were particularly interested in looking at performance of stands 15-25 years after harvest.

This paper reports the extension of a study that was presented in the Prescribed Fire in the Intermountain Region Symposium Proceedings (Bosworth 1989). The 1989 paper discussed differences in height growth found on clearcuts that were logged in the years 1960 to 1965. Since then the study has been extended to include clearcuts that were logged in 1966, 1967, and 1968.

METHODS

District personnel monitor the progress of stands after harvest with a regeneration status survey. On a plot, regeneration is tallied by species, height to the nearest foot, and last year's leader growth to the nearest inch, and trees are classified "crop" or "excess." Relatively high numbers of plots are surveyed, since one of our main concerns early in the rotation is poor distribution of regeneration. The high number of plots monitored was important to this study, as this gave us many trees to assess. Plots are systematically distributed throughout the clearcuts on a grid basis.

The stand database was queried for all clearcuts harvested during the period of interest. Stands were selected

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for analysis if both tree height and leader growth estimates were recorded for the plots. A total of 104 stands had adequate data from 2,752 plots.

Stands were stratified by site preparation method; 58 were treated by broadcast burning, 38 by bulldozer piling, and eight had no site preparation.

Each regeneration status plot in a selected stand was analyzed. The dominant tree was selected, and species, total height, and last year's leader growth of that tree were recorded. Since the stands were on differing inventory schedules, and year of last examination varied, it was necessary to normalize the data for analysis. For phase 1, 1960-65, of the study trees on plots examined in years prior to 1982 were "grown" to 1982 by adding appropriate numbers of annual leader growths to the total height estimate at time of exam. For phase 2, 1966-68, a similar process was applied for trees on plots examined prior to 1986. For comparison between the two phases of this study, years 1963/1966; 1964/1967; and 1965/1968 had been clearcut for the same number of years at time of analysis.

A mean total height (1982 basis, phase 1 and 1986 basis, phase 2) and mean leader growth were computed for each site preparation treatment for each year of clearcut origin. Analysis of variance was then used to compare treatments to see if differences in the means were related to treatment (Freese 1967).

RESULTS AND DISCUSSION

In phase 1, we determined that, in general, bulldozer piling for site preparation had a negative impact on both height growth and total heights of trees, when compared with broadcast burning and no site preparation. From the analysis of phase 2 data, it is apparent that these same trends are still evident, although there is some evidence that the differences between treatments are not as great in the newer stands.

Figure 1 shows average leader growth by year of harvest. In 8 years out of 9, growth on burned sites exceeded growth on bulldozer-piled sites. Growth on burned sites also exceeded that on nonprepared sites in 7 years out of 9. These differences are related to treatment and are significant at the 0.01 level in all years except 1962. In 1962 there is no significant difference in growth between burned and piled sites, but the differences between grow- th on nonprepared sites are significant when compared with the other treatments.

Where leader growth differences were significant, growth on burned sites was from 1.6 to 5.2 inches per tree better than on piled sites, and from 1.7 to 2.0 inches better than on nonprepared sites in years where burned areas had the best growth. When growths were best on nonprepared sites (1964, 1967) they exceeded growths on burned sites by 0.6 and 1.4 inches per tree.

When average total heights are compared, in 7 of 9 years, trees on burned sites are taller than trees on bulldozer-piled sites (fig. 2). The exceptions are years 1962 and 1966. In the 5 years when all three treatments occurred, burned sites had the tallest trees twice, nonprepared sites had the tallest trees twice, and bulldozer-piled sites had the tallest trees once. Analysis of variance

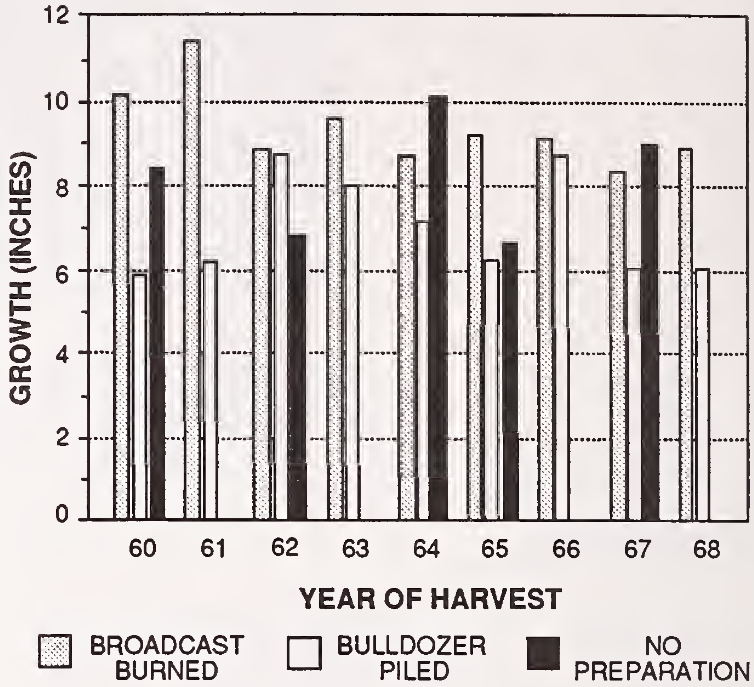

Figure 1—Mean growth of crop trees; comparison of site preparation methods.

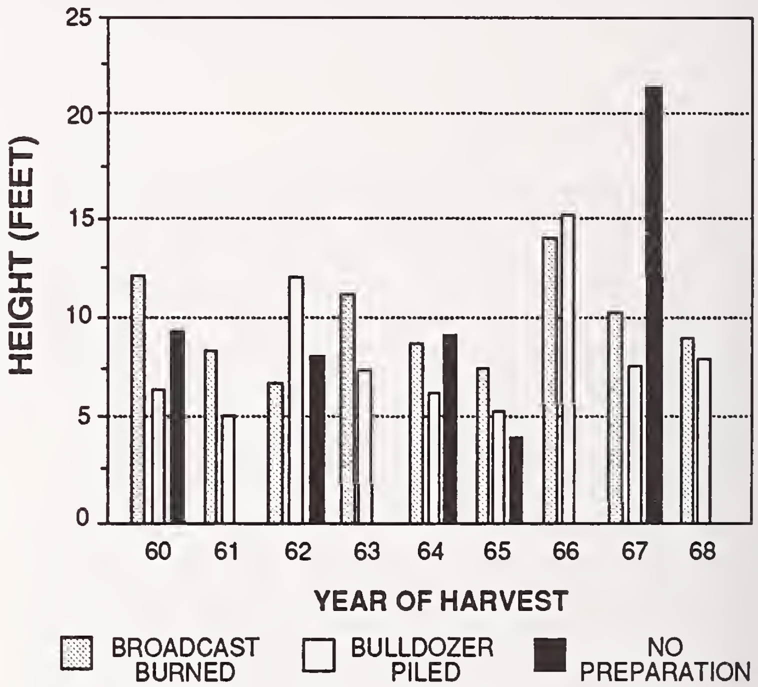

Figure 2—Mean height of crop trees; comparison of site preparation methods.

shows that the differences between treatments for all 9 years is significant at the 0.01 level.

In the 7 years where average heights on burned sites exceeded the heights on piled sites, the differences in means ranged from 0.9 foot in 1968 to 5.7 feet in 1960, with an average 2.9-foot height advantage on burned sites. In the 2 years when piled sites had the taller trees, the differences were 5.3 and 1.3 feet.

Caution is advised in drawing conclusions about the total height differences expressed by the nonprepared

site data. Examination of the trees included in these plots showed that they include some very old (70-year) advanced regeneration, and this factor is accentuated by the fact that in years where the differences are greatest, there is a very small sample. This advanced regeneration was not removed from the study because it is possible that some old advanced regeneration may also have survived piling or broadcast burning and may be in those plots also. Their effect is less in the pile or burn plots because there are generally many more trees in this part of the analysis, and a few trees do not affect the average as much.

The implications of the differences in height growth displayed in figure 1 in favor of burned sites over bulldozer piled sites, combined with the differences in total height displayed in figure 2, generally in favor of burned sites, cannot be ignored. A total of 94 stands comparing 2,496 trees are represented. Smith (1962) stated, "Within a stand, few trees ever recover a dominant position after they have fallen behind in the race for the sky." The trees in this study on the bulldozer-piled sites have not lost their dominant position in the stand, but even so, on the average, they are falling behind in height growth when compared with trees on burned sites. Based just on differences in total height, with no allowance for differences in diameter growth, if this trend continues for the remaining four-fifths of the rotation, the trees on the burned sites will average 10 to 22 percent greater sawlog volume than those on piled sites. It is reasonable to expect that if factors are reducing height growth of trees, they would also be adversely affecting diameter growth, thus producing even greater volume differences between treatments.

There are several reasons that these differences in height growth are being observed. One possibility is the flush of nutrients available for plant growth following broadcast burning (Holdorf 1982). There are also at least two possibilities for the reduced growth on piled sites. Compaction of soils has been shown to reduce height growth (Froehlich 1978). Even with one or two passes of a tractor over the soil while piling, some increase in soil bulk density occurs, particularly if the soils are wet. Another, and perhaps more important, reason may be related to nitrogen fixation. Nitrogen-fixing activity is normally between 5 and 10 times greater in organic versus soil horizons (Jurgensen and others 1980). It is not uncommon for much, if not all, of the litter and humus layers to be shoved into the piles during bulldozer site preparation. This is especially true when high volumes of logging debris exist. In that case, it is very difficult to keep from scraping all the organic matter off the site. This would greatly reduce the soil nitrogen-fixing process over much of the area piled.

It is likely that the differences in height growth demonstrated in this study are the result of a combination of these factors. The design of this study was not sufficient to ferret out precisely what the causes of the differences were.

Results of the phase 2 analysis are encouraging from several aspects. First, differences in height growth on burned and piled sites are not as great as they were in phase 1. This indicates that management decisions to use brush blades versus straight bulldozer blades, and to not pile slash as thoroughly, were in fact good decisions and are resulting in less impacts to the site.

The second encouragement from phase 2 is the indication that newer work is producing better trees than older work. Phase 1 trees-year 1963 are the same age as phase 2-year 1966; year 1964 the same age as 1967; and year 1965 the same age as year 1968. In figure 2, it is apparent that newer work is producing taller trees at younger ages in both the burning and the piling treatments. Some factors that may be causing these improved tree heights may be: (1) improving coordination to get prompt planting accomplished following site prepartation; (2) improving planting stock handling and storage prior to planting; (3) improving attention to detail of properly planting each tree; (4) decreasing site preparation effects (less intense broadcast burns and less site disturbance during piling operations); and (5) earlier reaction to scattered mortality in plantations by interplanting.

CONCLUSIONS

These data indicate that trees on broadcast-burned sites will have a height growth advantage over trees growing on bulldozer-piled sites. This advantage will be evident for at least 20 years. If the manager has a choice of methods, burning should be chosen over piling.

However, in situations where there is a big program, the manager needs the flexibility to use both methods in order to accomplish timely reforestation. Consequently, when piling must be used, it is important that everyone involved in the process be aware of the potential for long-term growth losses on those sites. The logger, the contract administrator, and the bulldozer operator need to know the effects of excessive scraping of the organic layers of the soil and also the potential for compaction of the site from "overworking" the area with a tractor.

In addition, silviculturists and fuel managers must take a critical look at some stands to determine if site preparation is in fact needed. Unnecessary working of a site by heavy equipment may cause hidden long-term damage.

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Speakers answered questions from the audience after their presentations. Following are the questions and answers on this topic:

Q. (from Jeff Collins)—Your graph indicates that height growth is equal or greater on nonprepared sites compared to bulldozer piling. Would you consider not piling if you could not burn?

A.—Yes, I would consider no treatment, if neither site preparation or hazard reduction are needed to meet objectives. Caution is warranted in placing strong interpretation on the nonprepped sites, due to the very small number of trees in those samples and the fact that inspection of those data shows that many of those trees are very old (70 year) advance regeneration.

Q. We heard earlier that tree height may not be a good indicator of fertility response in conifers. How might this observation affect the outcome of your study results?

A.—In trees the age that were studied, growth as measured in leader growth is simple to estimate. This leader growth indicates the sum of all factors that are affecting the tree. One uniqueness of this study is that it involves many trees, in many stands, on many sites, from several years of treatment. These trees are demonstrating differences that are related to the different treatments. As I stated in my conclusions, we do not know precisely why these differences in growth are demonstrated, but we do know that it is related to piled versus burned sites. The reduction in height growth on piled sites, regardless of the reason, cannot be ignored. It does seem very probable that nutrient availability is a part of the differences shown, given our current understanding of the importance of organic material to forest soil nutrients and knowing the extent of scraping off of organic material in the bulldozer piling that was typical during the 1960's.

THE GREATER YELLOWSTONE AREA FIRES OF 1988: RECOVERY

Henry F. Shovic

ABSTRACT

The Greater Yellowstone Area (GYA) fires of 1988 were of unprecedented scope and magnitude. Short- and long-term effects of these fires are presented for watershed, soils, and revegetation. Though there has been short-term erosion and sedimentation, overall there are probably insignificant long-term effects in the GYA.

INTRODUCTION

The Greater Yellowstone Area (GYA) is made up of parts of six National Forests and two National Parks, and assorted State and private lands. The portion in Federal ownership is about 11.7 million acres, within three States—Montana, Idaho, and Wyoming. Much of the area is nationally recognized recreation country. Wildlife resources and habitats are highly valued, both in terms of the species mix and in the great diversity of environments. Water resources include the headwaters of five major river systems, with high water quality and premium fish habitat. There are timber and mineral resources on National Forest and private lands. The scenery is unparalleled, ranging from the Tetons, through the rolling wilderness of Yellowstone, to the jagged peaks of the Absaroka range in the Gallatin and Custer Forests. The largest and most active hydrothermal areas in the world are in Yellowstone Park. Many people work in the GYA in jobs ranging from timber production to tourism. All in all, it is wonderful place to work, play, and live.

However, not all was perfect in paradise in 1988. As is usual in the GYA, some forest fires started in June and July. These fires, however, grew to sizes that were not usual. Fire behavior exceeded all predictions as July, August, and September came and went. Wind, drought, and vegetation types combined to produce unheard of fire behavior. Nighttime flames were more than 10 feet high, as winds drove fires through willows and other normally moist vegetation. Canopy fires made daily runs measured in miles, and surface fires burned through forests that normally are highly resistant to burning. The fires threatened almost all developed areas in Yellowstone Park and communities bordering Yellowstone and the Gallatin, Custer, Shoshone, and Targhee Forests.

Some unusual humor was generated during this intense activity. "Cooke City" signs were modified to read "Cooked City." The military was once again called in to help, as they were in the beginning days of Yellowstone Park. Over 4,000 military personnel helped the thousands of firefighters already committed to suppression and rehabilitation.

When the flames were finished, large areas looked like a blackened wasteland. The media and others called it "a biological desert," "a moonscape," a "destroyed ecology." But was that true? Is it true today, a year after? My purpose today is to talk about what really happened in the GYA during the fires of 1988, and what is happening today. I will speak as a soil scientist, but also as a participant in the greatest and most visible fire suppression and recovery efforts this Country has ever seen. My information comes from many sources. Part comes from personal experience and research, but much comes from the many specialists (both Forest Service and Park Service) and scientists working in the GYA. These people have been working hard on postfire research and monitoring to address the effects, and I appreciate their willingness to give me the data I needed to address the overall picture for you.

COORDINATION AND ROLES

The various administrative units have created a committee charged with increasing coordination and communication between them, called the Greater Yellowstone Coordinating Committee (GYCC). It is made up of six National Forest Supervisors, two National Park Superintendents, three Forest Service Regional Foresters, and one Park Service Regional Director. As part of its coordinating role, the committee completed the Greater Yellowstone Area Aggregation of Management Plans. Its objective is to illustrate the goals of and relationships between NationalPark and National Forest management, along with displaying the resource values within the GYA. The committee also has published the GYA Briefing Guide, a summary of resources, administration, management philosophies, and future directions.

Currently the GYCC is applying effort to draft a "vision for the future." The major challenge in this effort is to protect the values associated with the GYA and its various units, while continuing to provide products and services consistent with their management directives and legislated objectives. This vision is being developed in terms of Goals (the desired future conditions of the GYA), and Coordinating Criteria (the ways the Goals will be reached). Your input is solicited to help make this product the best possible one. After initial drafting and public input, the GYCC is producing a "draft vision" to be released for additional public comment in late spring of 1990. Then, a final document will be completed in late

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fall or early winter. The next steps are to compare existing management plans to these Goals, and make amendments and revisions to the plans as necessary.

My position in the GYA is as an interagency soil scientist, working both for the Forest Service and the Park Service. This position is in the true spirit of the GYA, as part of my job involves improving coordination, cooperation, and communication between the units in which I work. I have been asked to initiate a soil survey for Yellowstone National Park and for parts of Grand Teton National Park. I am doing project services, particularly in site restoration for Yellowstone and Grand Teton Parks. I provide soils input for Forest Plan implementation activities for the Gallatin Forest. I have also acted as an informal "GYA" soil scientist during the fires (and now, for that matter), doing fire/soil interpretations, burned-area mapping, and rehabilitation. I held the first GYA soil scientist meeting in 1986, and we are repeating that meeting this year. Giving soils tours has often brought managers together for informal talks that otherwise may not have occurred.

So what did I do during the fires of 1988? In the early part of the summer I did what most soil scientists do; that is, dig soil pits in out-of-the-way places. I ran the soil/watershed/geothermal group in the Division of Research at Yellowstone and did numerous projects for the Gallatin Forest. However, the fires changed all that. What looked like snow on buildings at Old Faithful was actually a blanket of fire retardant as buildings were foamed down to protect them from the flames. This represents my change in orientation as I left the soil survey to become responsible for numerous projects, such as burned area mapping, fire/soil interpretations, emergency rehabilitation, initial fire/soil research, and information transfer. My team and I did infiltration and erosion tests using rainmaking equipment. We sampled for "depth of char" (depth of soil heating) in thousands of locations. We sampled well into the winter for the most important soil features that would affect revegetation and erosion.

FIRE EXTENT AND SEVERITY

So what were the effects in the GYA? Was the area a moonscape? Were the Forests and Parks destroyed forever? Were erosion and stream sedimentation going to be catastrophic? What rehabilitation was needed, and what good would it do? To answer these questions, I would like to begin by giving you a perspective on the fires' extent, and severity. Then I will deal with the immediate effects and finally talk about the long-range ones.

"How much burned" was a critical question. To answer that, I undertook an emergency project to map the extent, severity, and distribution of the fires. This was completed in 3 weeks in late September of 1988. The project, sponsored by the GYCC, was an interagency one, with cooperation from the Forest Service, Park Service, NASA, and other agencies. Acreage burned is given in the document "Preliminary Burned Area Survey of Yellowstone Park and Surrounding National Forests." A map is included, and the information is available on USGS quadrangles and in an electronic format. This was a reconnaissance survey, designed for initial predictions and assessment of rehabilitation needs.

We completed a more detailed survey for Yellowstone Park in December of 1989, again emphasizing interagency cooperation, and mapping to a finer scale. This is titled "Burned Area Survey of Yellowstone National Park." The mapping base was a combination of LANDSAT imagery and digital information from aerial photographs. This was done to provide Yellowstone management with the detailed information needed to predict effects on its resources and as a research aid. Completion of this project would not have been possible without the use of a Geographic Information System, purchased by Yellowstone Park after the fires.

Predictions were given on revegetation and erosion potential for both projects. Though fires were of spectacular intensity, actual severity of soil heating was low to moderate throughout the GYA. Depth of char ranged from 0 to 4 cm. We predicted that erosion, stream sedimentation, and effects on fish habitat would not be significant, given a normal water year and the emergency rehabilitation efforts accomplished. Also, revegetation would not be delayed due to soil damage. Burned areas in the Shoshone National Forest, however, had more severe soil effects, due to topography, weather, and vegetation. These differences affected revegetation, erosion, and stream sedimentation. Predictions for the Shoshone were for more severe effects.

REHABILITATION WORK AND RESULTS

Now I would like to give you a unit-by-unit description of fire effects and major rehabilitation efforts. Some units, like the Bridger, carried out only a small program of rehabilitation. Others required a considerable effort. This was because of the variation in resource loss potential. I will speak about National Forest lands first.

National Forest Lands

Forest Service philosophy in emergency rehabilitation is to protect resource values. Emergency measures are taken when there is significant potential for loss of soil and onsite productivity, loss of water control and deterioration of water quality, or threats to life and property on- or offsite. For the GYA Forests, efforts were based on the Preliminary Burned Area Survey and additional field investigations.

The Gallatin and Custer Forests had about 126,400 acres of burned area, all in Wilderness. Rehabilitation efforts were coordinated by the Gallatin. About 4,000 acres were seeded with cereal rye, a short-lived, vigorous grass. Streamsides were seeded because fires had burned to the water's edge, and rapid vegetative recovery was needed to prevent excessive sediment reaching streams in a short period of time. One hundred and fifty miles of trails were cleared and waterbars were installed. We needed this to access the areas and to prevent trails from becoming gullies or stream channels as bare slopes shed rainfall. Log erosion barriers (LEB's) were installed on

400 acres of steep, burned area above Cooke City and Silver Gate to reduce rill erosion and encourage infiltration. Postfire monitoring indicated that the cereal rye provided a good surface cover where seeded in blackened areas. Unburned meadows had very little competition from the rye. Waterbars were effective, but now will need maintenance to keep them functional until slopes revegetate. Slopes with installed LEB's had very little slope wash and were covered with a dense growth of rye. Future erosion should be minimal, given normal precipitation patterns.

The Targhee National Forest had about 11,300 acres of burned area. Most of this area was on rolling or gently sloping land. Seeding on 800 acres of steep, burned slopes was moderately successful in germination. About 60 miles of bulldozer lines were seeded also. Though the burn had no measurable or lasting effects on water quality to date, there were significant local erosional events because of a series of intense thunderstorms that moved through the area in August. These produced significant erosion and stream channel scour in Moose Creek, Thirsty Creek, and Lucky Dog Creek.

The Bridger-Teton Forest had about 157,700 acres of burned area. This was all in Wilderness, and no seeding was recommended. A few trails were cleared. There has not been significant sedimentation in this area to date. There is some potential for localized erosion on steep slopes if there are unusual precipitation events.

The Shoshone National Forest had about 118,800 acres of burned area. Here, soil heating was more intense than in other areas, probably due to higher fuel loadings, canyon topography, and weather. Depth of char was 5-10 cm in many areas, and revegetation is delayed due to charring of belowground plant parts. Water repellency persisted through the winter, and many areas still have this property. The Shoshone Forest helicopter seeded 23,000 acres with a native seed mix. Trails were cleared and waterbars installed. Though these measures undoubtedly reduced some erosion, the severe thunderstorm events of August produced major channel scour, debris torrents, and rill erosion largely from slopes in the Jones Creek and Lodgepole Creek drainages. Postfire monitoring indicates that water repellency is persisting, vegetative recovery is variable, and erosion will probably continue for at least 5 years. Further seeding is not recommended, but extensive trail maintenance is needed to prevent further erosion.

National Park Lands

National Park lands are managed under different constraints than Forest lands. They are managed as a "natural" system, at least as far as feasible. Their directives dictate the preservation of the natural processes, while providing for public use. Protection of developed areas, visitors, and downstream uses is considered in rehabilitation decisions, as well as cost-effectiveness and feasibility.

About 773,000 acres in Yellowstone Park were affected by fire. Soil heating was quite uniform across the burned areas. Almost all the area had moderate-to-low soil heating, and predictions of revegetation were optimistic. There were some unusual soil effects in hydrothermal areas, such as where molten sulfur flows occurred due to ignition by fire brands.

To date, revegetation has been spectacular, with fields of wildflowers and grasses replacing blackened soils. Revegetation is not as obvious at higher elevations, as there are less forbs and grasses, shorter growing seasons, and more shade.

We were fortunate to have an ongoing sedimentation/turbidity study in place for 3 years before 1988. Results for 1988 showed no measurable increase in sedimentation from the fires, at least in the northern third of the Park. However, there were days when some streams ran black as dissolved ash entered the system. These events did not significantly affect fisheries. Water repellency in soils was high both on burned and unburned areas, probably due to the extreme dryness of the soils during the drought. This water repellency largely disappeared over the winter of 1988-89. Erosion has been minimal, with the following exceptions.

In August, a series of intense thunderstorms created debris torrents near Gibbon Falls and Madison Junction. I involved Forest Service and USGS research personnel in an evaluation of these events. Reports indicate these torrents were caused by a combination of steep slopes, very coarse-textured soils, intense rainfall, and bare surfaces due to low vegetation cover and the fires. These are localized events, and eroded material did not significantly affect fisheries. Road maintenance and safety were problems for a few weeks. There were no significant effects on thermal features other than the sulfur flows.

Park Service officials authorized some logging along roadsides to prevent logfalls on roadways, and to improve esthetics of the road corridors. Forty-one miles of bulldozer lines were made near West Yellowstone and Cooke City as part of suppression efforts. About 39 miles of bulldozer line have been restored. We returned topsoil, brush, and logs to the sites. Emphasis was on restoring original site characteristics to as close to the undisturbed condition as possible. Five hundred of a total of 575 miles of hand-built fireline have also been rehabilitated. However, more hand line is probably present in the backcountry, and will be restored when located.

Visitor education has been emphasized. Soil/vegetation/fire exhibits have been set up throughout the Park. I have made numerous field trips and presentations for visitors and staff. We have also done winter field trips to show depth of char and burn patterns.

Grand Teton Park had only a small area affected by fire (2,700 acres in the John D. Rockefeller Parkway). However, there is potential for more fires in the future, particularly in the northern part of the Park. There were other small fires in this unit, but they were of small extent and had no significant effects on soil/water resources. Extent of soil heating and revegetation are similar to those discussed earlier. As part of the postfire research, I am initiating a soil/vegetation survey of the Parkway. Dick Marston at the University of Wyoming has done some postfire work on channel changes in the Snake River and on soil erosion. His preliminary findings indicate that soil erosion potential has increased somewhat, but is unlikely to result in catastrophic erosion on slopes.

He has indicated that no channel changes were detectable as a result of the fires, but there are probably significant changes as a long-term result of the Jackson Lake dam.

Rehabilitation efforts in the Grand Teton Park emphasize visitor education. I have given soil/revegetation training for Park staff.

SUMMARY

In summary, you can see there have been short-term effects of the fires of 1988. The most obvious is that the area looks different. Though this is the most apparent, I have shown today that there are also effects in revegetation, erosion, and short-term sedimentation. This information is a direct result of postfire monitoring and research in the units of the GYA. The erosional events were generally localized and resulted from unusual precipitation occurrences. Some areas will produce significant sediment for some time, but the vast majority of lands have little short-term erosion potential due to the moderate fire severity and rehabilitation efforts. Early-stage vegetation succession is proceeding as predicted. In 10 years, the blackened soil will be gone as microorganisms, frost, and plants do their work.

In the long run, these fires probably have had insignificant impacts on the area. In fact, they are an essential part of the GYA ecology. The present vegetation patterns are partially a result of past fires that return at long intervals. The stream systems, the fish, the wildlife, and soils have evolved in harmony with these events and vary as they always have in response to disturbances, like the fires. If I can leave you with a summary thought, it is that after the smoke has cleared, the GYA is alive and well. Its impressive scenery, wildlife, timber, and other resources are, in the long run, left unimpaired for future generations to conserve and preserve in the spirit of the Greater Yellowstone.

FACTORS AFFECTING ECTOMYCORRHIZAE AND FOREST REGENERATION FOLLOWING DISTURBANCE IN THE PACIFIC NORTHWEST

Michael P. Amaranthus

ABSTRACT

Studies from the Pacific Northwest indicate that disturbance can reduce ectomycorrhizal formation and forest regeneration. However, the degrees of reduction and impact on forest regeneration vary widely and depend on many factors. Among these are the type and severity of disturbance, ectomycorrhizal diversity, climatic conditions, biotic conditions, and the effects of nonhosts over time. Mycorrhizal formation and regeneration are most greatly impacted on severely disturbed and environmentally limited sites. The rapid occupation of such sites by ectomycorrhizal host plants following disturbance is critical to stabilizing native mycorrhizal populations that may aid forest regeneration.

INTRODUCTION

Much of the research on effects of forest disturbance on soil biology has focused on ectomycorrhizae. Because most forest-tree species in the Pacific Northwest require ectomycorrhizae for nutrient and water uptake, the importance of understanding the relationship between disturbance, site conditions, and mycorrhizae cannot be overstated. Numerous authors have reported reductions in mycorrhiza populations due to forest disturbance (Amaranthus and others 1987; Harvey and others 1976, 1980; Parke 1984; Perry and others 1982). However, the degree of reduction and its impact on forest regeneration varies widely and depends on many factors.

TYPE AND SEVERITY OF DISTURBANCE

Timber harvest and site preparation are the most widespread forest activities in the Pacific Northwest that alter both the aboveground and belowground environments and therefore potentially impact ectomycorrhizae. In the Pacific Northwest, clearcutting and prescribed burning are the common harvesting and site preparation practices. Soil nutrient status, moisture, temperature, pH, and organic matter contents, litter inputs, and species composition affect the community structure of soil organisms (Harvey and others 1980)—and all of these are influenced by harvesting and site preparation. Clearing vegetation and disturbing the forest floor remove nutrients and reallocate them within the ecosystem. Harvesting host trees eliminates the photosynthate source for dependent ectomycorrhizal fungi and associated microbes. Converting a mature forest to a clearcut increases soil temperatures by removing the protective canopy. Prescribed broadcast burning increases soil pH, creates a nutrient flush, and can reduce litter and duff levels (Amaranthus and McNabb 1984). Soil organic matter, humified material, and decaying wood are centers of microbial activity and are substantially diminished as a result of intense fire. Changes in aboveground community composition alter the quality and quantity of root exudates and litter leachates.

Wright and Tarrant (1958) found fewer ectomycorrhizae on Douglas-fir seedlings growing in burned, compared to unburned, clearcuts. The greatest reductions were associated with the hottest burns. Thus, not only the type of activity, but its severity, is critical. Parke and others (1984) compared mycorrhiza formation in soils from burned and unburned clearcuts of 36 "difficult to regenerate" sites in northwestern California and southwestern Oregon. Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) and ponderosa pine (*Pinus ponderosa* Laws.) seedlings grown in soils from the burned clearcuts formed 40 percent fewer ectomycorrhizae and seedlings grown in soils from the unburned clearcuts 20 percent fewer ectomycorrhizae than seedlings grown in undisturbed forest soil. Amaranthus and others (1987) found 90 percent less native mycorrhizae and 44 percent less basal area growth on Douglas-fir seedlings grown in clearcut and severely burned soils compared to undisturbed forest soil. However, all studies do not report mycorrhizal reductions following clearcutting and prescribed fire (Pilz and Perry 1984; Schoenberger and Perry 1982). It is difficult to generalize about effects of burning on ectomycorrhizal populations because they are highly dependent on duration and intensity of fire as well as soil and site conditions (Perry and Rose 1983).

Because ectomycorrhizae predominate in the organic layers of the soil (Harvey and others 1976, 1979; Trappe and Fogel 1977), the degree of organic matter lost from a site can influence mycorrhiza populations. Harvey and others (1976) found up to 95 percent of the active ectomycorrhizal fungus types in humus and decaying wood in a mature Douglas-fir/western larch (*Larix occidentalis* Nutt.) forest. The importance of decaying wood to support ectomycorrhizal populations may be most

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critical following disturbance such as wildfire. Immediately following the 1987 wildfires in southern Oregon and northern California, decaying wood contained 25 times more moisture compared to mineral soil and was a center of ectomycorrhizal activity for recovering vegetation (Amaranthus and others 1989). Decaying wood also acts as habitat for small mammals that are important in distributing fungal spores of several belowground mycorrhizal fungi (Maser and others 1978).

ECTOMYCORRHIZAL DIVERSITY

Ectomycorrhizal diversity within the soil may buffer the impact of disturbance on forest sites. Disturbance did not reduce ectomycorrhiza formation on Douglas-fir seedlings grown in soil from extremely productive sites in the Oregon Cascades, where diversity of ectomycorrhizal fungi is high (Pilz and Perry 1984; Schoenberger and Perry 1982). However, the proportion of each mycorrhizal fungus type shifted significantly with both soil disturbance and plant community; different environments reduced ectomycorrhiza formation by some fungus types and apparently stimulated formation for others. In contrast, clearcutting significantly lowered ectomycorrhiza formation on Douglas-fir seedlings grown in soil from a harsh, less productive site in southwestern Oregon, where soil contained relatively few ectomycorrhiza types (Amaranthus and others 1987). Reduced ectomycorrhiza formation correlated positively with decreased basal area Douglas-fir growth after outplanting. The southwestern Oregon soil, with low fungus diversity, was poorly buffered against disturbance, compared to the soil from the Oregon Cascades.

CLIMATIC CONDITIONS

Climate influences seedling growth and ectomycorrhiza formation (Harvey and others 1980; Pilz and Perry 1984). The importance of early mycorrhiza formation in dry areas has been emphasized (Mikola 1970; Parry 1953). Dry climates may limit the activity of mycorrhizal fungi by decreasing the length of time for spore production, germination, and optimal mycelial growth, which in turn can decrease the chances for planted seedlings to become colonized (Amaranthus and Perry 1987). Seedlings in moist climates may be able to survive longer without mycorrhizae than those in dry climates, increasing their chances of becoming colonized. Moisture content also affects uptake of certain nutrients by mycorrhizae (Gadgil 1972).

Seedlings growing in cold climates may also require rapid, early mycorrhizal colonization to take advantage of the short growing season and obtain the necessary nutrients and water to survive the long cold season and early frosts. In studies in the Klamath Mountains of northwestern California and southwestern Oregon, Amaranthus and Perry (1987, 1989a, 1989b) found that mycorrhiza formation most strongly influences seedling survival and growth on sites limited by both moisture and temperature.

BIOTIC CONDITIONS

The importance of aboveground community structure to belowground biological functioning is unclear. It is increasingly apparent, however, that an ectomycorrhizal fungus can link some plant species together through shared fungal mycelia (Bjorkman 1970; Finlay and Read 1986; Read and others 1985). In the natural forest environment, ectomycorrhizal fungi, supported by non-coniferous hosts, can actively colonize conifer seedlings. Root-chamber analysis of the development of ectomycorrhizal mycelium has shown that expanding hyphal fans not only act as nutrient-absorbing structures but also can colonize nonmycorrhizal feeder roots in host-plant combinations within and among host species. Using radioactive labeling, Finlay and Read (1986) have demonstrated the movement of carbon among plants connected by mycorrhizal mycelia. Clearly, the existence of "pipelines" for distributing materials among plant species has important implications for forest regeneration following disturbance.

Little is known of the persistence and distribution of ectomycorrhizae in the absence of living hosts. It has been suggested (Hacskaylo 1973) that ectomycorrhizal fungi do not persist long in the absence of host-supplied substrate. In the Klamath Mountains of northwestern California and southwestern Oregon, sites that have been logged and burned are often rapidly invaded by woody shrubs (Gratkowski 1961). These shrubs, members of the Ericacae and Fagaceae, form mycorrhizae with many of the same fungi as do members of the Pinaceae (Molina and Trappe 1982). These shrubs may preserve mycorrhiza diversity during periods of rapidly changing aboveground community structure.

Amaranthus and Perry (1989b) planted Douglas-fir seedlings at two locales: (1) a site cleared of whiteleaf manzanita (*Arctostaphylos viscidi* Parry); and (2) a meadow cleared of annual grasses. In the first year, there was more dramatically improved seedling survival and growth on the manzanita site than the adjacent meadow with similar moisture and temperature conditions. By the second year, if basal area growth of surviving individuals is taken into account, Douglas-fir at the manzanita site, on an area basis, was nearly 10 times that of seedlings in the cleared meadow. Douglas-fir seedlings, outplanted at the manzanita site, formed mycorrhizae more rapidly than those seedlings outplanted at the meadow. There were also dramatic shifts in the types of mycorrhizae found on seedlings grown at the two sites. Douglas-fir seedlings, at the manzanita plots, contained significantly higher proportions of *Rhizopogon* sp. mycorrhizae. Certain *Rhizopogon* species have been demonstrated to decrease seedling moisture stress and improve seedling outplanted performance. Rapid mycorrhiza formation, with fungi well adapted to site conditions, is key to seedling establishment on sites difficult to regenerate.

Some woody shrub species may act as biological reservoirs, not only of mycorrhizal fungi but of other microflora as well. Significantly higher rates of nitrogen fixation—and increased seedling survival and growth—were found

in association with the mycorrhizae of Douglas-fir seedlings in a stand cleared of whiteleaf manzanita than in a meadow cleared of annual grass (Amaranthus and others 1990). *Azospirillium,* a nitrogen-fixing bacterium, was isolated with Douglas-fir mycorrhizae at the manzanita site. Whiteleaf manzanita occupies particularly hot, dry sites where fire is frequent. Because high nitrogen losses can accompany intense fire, natural mechanisms by which nitrogen is returned to the soil can be important to forest regeneration.

EFFECT OF NONHOSTS OVER TIME

Many grass and shrub species, such as *Ceanothus* and *Rubus* (for example, blackberry, salmonberry), form vesicular-arbuscular mycorrhizae (VAM) with fungi incompatible with members of the Pinaceae (Rose and Youngberg 1981; J. M. Trappe, unpublished data, USDA Forest Service, Pacific Northwest Research Station). On sites long dominated by VAM species, the ectomycorrhizal fungi needed by members of the Pinaceae will gradually diminish, and the soil microbial complex associated with ectomycorrhizae can be reduced. Invasion of sites by nonectomycorrhizal plants over years can seriously affect reforestation (Amaranthus and Perry 1987), particularly in the case of ectomycorrhizal tree species growing on difficult sites where seedlings must establish ectomycorrhizae early to survive.

How long soils retain their mycorrhizal colonization potential in the absence of living hosts is unknown. Ectomycorrhizal spores and hyphal fragments have remained metabolically active after 2 years in Scandinavian forests, though the number of active fragments dropped dramatically over that period (Ferrier and Alexander 1985; Persson 1982). In the Pacific Northwest, mycorrhiza formation generally decreases as the length of time between disturbance and reforestation increases (Perry and others 1989; Pilz and Perry 1984).

CONCLUSIONS

Studies from the Pacific Northwest indicate that reductions in mycorrhizal formation may affect the performance of outplanted seedlings, particularly on severely disturbed or environmentally limited sites where rapid early growth is important. On these sites, planting seedlings with well-developed mycorrhizae and protecting native populations of mycorrhizal fungi aid regeneration. Why mycorrhizal formation is reduced following disturbance in some areas and not others is not understood. Great variability in the physical and biotic environment and disturbance history and intensity are likely contributing factors. One factor that appears to be important is the presence of ectomycorrhizal host plants. On many sites, noncommercial plants may serve as reservoirs of ectomycorrhizal fungal inocula while conifers are becoming established.

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MINIMIZING THE ADVERSE IMPACTS OF TIMBER HARVEST IN THE NORTHERN ROCKY MOUNTAINS

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ABSTRACT

A new model, NUTROSS, has been developed to evaluate nutrient losses from harvest of aboveground biomass of trees in the Northern Rocky Mountains. The model is useful in low-precipitation forest zones with little or no solution losses to streams. It is based on the assumption that materials not removed in harvest will be available to supply nutrients through mineralization to grow the same ecosystem components in the next rotation. Those components of the stand that are removed in harvest should not contain more than 25 to 30 percent of any one biologically essential nutrient. These values are guidelines based on estimated weathering rates. The model was applied to seven stands that were thinned to 3 by 3, 4 by 4 ,and 6 by 6 m with unthinned controls. The results showed that available copper and zinc were most often inadequate to support intensive harvest of trees over three or four rotations. Most of the soils studied can support harvest of boles only.

INTRODUCTION

A chemically fragile soil is one that does not hold enough of one or more biologically essential nutrients in available (soil) or recyclable form (slash, ground vegetation, and litter) to support the next three or four forest rotations. Four rotations are reasonable for judging extremely poor soils, and three rotations are used for evaluating more fertile, warmer soils where weathering is more rapid. Studies on nutrient loss related to harvest (Stark 1982) showed that there should be enough nutrient onsite in available or recyclable form at the time of harvest to grow trees and associated vegetation for the next three to four rotations (70- to 100-year rotations). Growth during the next three to four rotations should not be subsidized from nutrients recently weathered that could otherwise allow maturation of the soil.

Most of the more productive forests have ample nutrients to support conventional harvest of boles to a 7-cm top on a 70-year rotation. Only those sites that have more than 25 to 30 percent of any one nutrient tied up in the biomass to be harvested are chemically fragile and need special management. Chemically fragile soils can be recognized by poor tree growth where there is adequate rainfall, or through chemical analysis. Chemically fragile soils have one or more of the following characteristics: shallow feeder root zone (<36 cm), or deeper but with high rock content (>60 percent rock), or are geologically young, or occur in either cold or dry areas where weathering is slow. Talus slopes are often chemically fragile as are shallow, cold, young rocky soils at high elevations.

A simple model called "NUTROSS" has been developed to help identify chemically fragile soils that cannot withstand conventional harvest (boles only) or intensive harvest (boles, branches, needles) in the more arid forest types.

NUTROSS, as it is perceived in the Northern Rocky Mountains, does not contain a nutrient export component for streamflow. The reason is that most areas with under 76 cm of annual precipitation on soils with an appreciable clay content do not normally lose large amounts of nutrients to streams after harvest unless erosion occurs (Stark 1982). The model assumes that no significant erosion will occur with logging or burning of slash. If erosion does occur, an erosion model is needed to predict nutrient losses. The soils in stands that can be analyzed by NUTROSS tend to have high infiltration but slow percolation rates and abundant vegetative cover. It is not unusual to find Montana forests with soils that are below field capacity at the lower extent of the feeder root zone on June 1. High-elevation forests with heavy snow packs contribute most heavily to streamflow in the Rockies. Nutrient losses to streams after logging at low elevations are usually insignificant (Stark 1982).

In Montana, most nitrogen fixation occurs in old logs, shrub or herb roots, or lichens that are not normally exported from the site. There may be some change in N fixation in logs after clearcutting because of increased soil temperature, but the nitrogen fixation system is left intact. If logging does not remove the nitrogen fixers themselves or their substrate, and there are minimal solution losses of N to streams, then there is no need to spend money estimating nitrogen loss. NUTROSS does not attempt to predict future tree growth. Until we can accurately predict future climates, atmospheric CO_2 and pollutants, disease and insect outbreaks, or mutations, it is

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risky to try to predict future tree growth. What we can do is to identify how much nutrient is available in the soil at the time of harvest and evaluate how many harvests can be supported by the available soil and litter nutrient pools, assuming minor climatic change. Since slash normally remains onsite, most of the nutrients in the slash should be released over 70 to 100 years in amounts adequate to supply those same components during the next rotation.

If sedimentary cycle nutrients do limit tree growth at the time of harvest, then the site is chemically fragile to that level of harvest and NUTROSS will show significant nutrient losses resulting from harvest. If nitrogen or other nutrients limit growth prior to harvest, foliar analysis will identify the problem before the model is applied. The inputs to the model require foliar analyses for micro- and macroelements. The standards for foliar analyses are in the frequency distribution table prepared by Zinke and Stangenberger (1979) or Stark (1981) or locally acceptable standards used to judge good growth for the species in question. If the foliage has a clear deficiency of a sedimentary cycle nutrient, then there is no need to run NUTROSS because we know that the soil does not have an adequate available nutrient pool to supply these nutrients for the next three to four rotations. The only ecologically sound alternative is not to harvest. If the foliage has sufficient sedimentary cycle nutrients but is deficient in N, harvest must be planned to minimize damage to the N-cycle by leaving large logs and more biomass onsite, or in extreme cases, aborted altogether.

NUTROSS is intended for sites where trees are limited in growth more by nutrients than water or cold soils, although water may limit growth late in the growing season. Xylem sap or foliar analyses that show deficient nutrient levels before spring flush when water is normally not limiting to tree growth (Stark and others 1985) are useful in evaluating the presence of nutrient stress.

With NUTROSS the user can specify any degree of harvest from a light thinning to clearcut by indicating a percentage of the total biomass to be removed and whether boles or whole trees would be cut. If a particular level of harvest predicts nutrient losses in excess of 25 to 30 percent, then the user can instruct the model to specify what level of harvest would result in acceptable or specified levels of nutrient loss. The final printout of NUTROSS displays what percentages of each nutrient (except nitrogen) will be removed with the proposed treatment.

These concepts are based on weathering studies in the Northern Rockies by Clayton (1984). They would change for other geographic areas and could be adapted to other semiarid forest ecosystems.

USING THE MODEL

There are some specific limitations to the use of NUTROSS in its present form. The model is recommended for the following conditions:

a. Moderately permeable soils in low-rainfall areas (<76 cm/yr).

b. Forested land that normally does not contain an appreciable nutrient export component for streamflow (before or after harvest).

c. Soils moderate in clay content (<40 percent), low in gravel or sand (<40 percent).

d. Forests with litter and duff accumulations <6-8 cm.

e. Forests with continuous understory vegetation or with species that sprout or reestablish quickly after disturbance.

f. Soils without sand lenses that "pipe" water.

g. Major nitrogen fixers that reside in rotten wood, shrubs, or lichens.

To use NUTROSS, the following must be known:

a. D.b.h. and height of timber present before and after harvest by species on ±10 sample plots per stand. Total stand volume present and projected for removal by species must be converted to weight per square meter using Brown (1978) or Faurot (1977).

b. Available nutrient content of the feeder root zone (subsamples composited within each profile by volume in 10-cm depth increments to the depth of the feeder root zone) for ±10 locations per stand.

c. Ten samples of the most recent needles from the lower third of the crown must be analyzed for the same 14 nutrients. These data in µg/g are examined for possible nutrient deficiencies. If severe nutrient deficiencies are identified (Zinke and Stangenberger 1979), and tree radial growth is poor (for Montana, >5 rings/cm) for the species and climate (because of nutrients), NUTROSS may not be needed because the site may not be suitable for tree harvest at any level. If there is a question about site potential in the absence of large trees, NUTROSS can be run using published data on the species where maximum and minimum nutrient concentrations (Stark 1981) allow calculation of the "worst and best possible case" scenarios.

d. The depth of the feeder root zone (95 percent of feeder roots) is measured in 10 soil pits per stand.

e. The percent coarse fragments (>2 mm), soil texture, litter depth, percolation through the B horizon, and annual precipitation are valuable to decide if the model is appropriate to a particular area.

The total recyclable nutrient load aboveground for each nutrient in the material to be harvested is converted to meq/m^2 and added to the available feeder root zone nutrients to represent the nutrient pool that could be lost. Those nutrients left behind in unharvested materials are assumed to remain onsite to grow the same components in the next rotation because of low leaching losses. If boles only are to be removed, then the nutrients in the boles represent presently removable nutrients. Nutrients (individually) in the bole to be removed are divided by the total of each nutrient in the potentially harvestable pool x 100, representing the percent nutrient loss from the ecosystem resulting from that type of harvest. Although nutrients in the needles and branches can be lost, the assumption is that they will not be lost because: (a) they will not be removed in harvest, and (b) once mineralized, the nutrients released from them into the soil are not

likely to be leached away because low precipitation and soil texture prevent a flush of the soil nutrients to streams.

If any nutrient to be removed during harvest exceeds 25-30 percent, the site cannot support three rotations in the future without subsidizing the nutrient pool from weathering, and that level is inappropriate for the site in question. Different harvest intensities can then be evaluated through the model to identify the harvest level suitable for the nutritional status of the particular site.

The NUTROSS model in its present form is:

RN_i = pecentage of ith nutrient lost or removed due to specified harvest of whole trees and optional burning

$$= \frac{(TNB_i - TNA_i - XN_i) \times 100}{(TNB_i + FN_i)}$$

or

RN_i = percentage of ith nutrient lost or removed due to harvesting of boles only

$$= \frac{(TNB_i - TNA_i - XN_i) \times 100}{FN_i}$$

where

TNB_i = pretreatment total nutrient i per square meter of surface area

TNA_i = posttreatment total nutrient i per square meter of surface area

XN_i = nutrient i lost below FRZ (feeder root zone) because of burning

FN_i = available nutrient i in FRZ per square meter of surface area

both TNB_i and $TNA_i = MN_i + WN_i + BN_i$ where

MN_i = total nutrient i in needles

WN_i = total nutrient i in wood and bark (>7 cm diameter)

BN_i = total nutrient i in branches (0-7 cm diameter)

i = 1, . . . , 8, or 13 nutrients

Then the amounts of each nutrient removed are compared to the amounts available as meq/m^2 in the litter and to the depth of the feeder root zone in the soil.

Since ecosystem nitrogen should be considered, the model is being modified to calculate how much of each nutrient would be left on the site if 16 metric tons of logs were left behind. Larsen and others (1978, 1979) determined that this amount of woody material is necessary to supply the organic habitat for nitrogen-fixing organisms and mycorrhizae needed for the uptake of a number of nutrients. Thus, 16 dry tons of wood/acre that would otherwise be harvested actually remain onsite, reducing the sedimentary nutrient drain from the harvest of boles and assuring reasonable habitat for mycorrhizae and nitrogen fixation.

FIELD TEST OF NUTROSS

Harvest studies at Lubrecht, 58 km east of Missoula, MT, resulted in the removal of different amounts of fiber and nutrients from several PSME (*Pseudotsuga menziesii*, Douglas-fir) habitat types. The soils are of varying geologic origin, and hence, of varying fertility. The stands were variable in stocking density before treatment. A question arose concerning the possible depletion of the available nutrient pool due to whole-tree harvest.

Seven stands of varying species composition were measured before harvest. The following four treatments were planned for each stand:

1. Control
2. 3- by 3-m spacing
3. 4- by 4-m spacing
4. 6- by 6-m spacing

Sampling was conducted as indicated earlier, and NUTROSS was run for eight nutrients.

Table 1 shows the projected percent of available ecosystem nutrients that would be lost if each stand had been clearcut (with boles only removed) rather than thinned. The treatments were not applied in this case, but the data were taken from plots within each stand before treatment. In theory, if the stand densities, tree sizes, ages, and soils were uniform, each treatment within a stand would show the same projected percent nutrient loss from clearcutting. Stands 1, 2, 5, and 7 show variability, especially for the projected losses of copper and zinc (table 1). Stands 5 and 7 are also highly variable and show high (>30 percent) projected nutrient losses from clearcutting with the removal of only the boles. If whole trees were to be removed, nutrient losses from stands 4 and 6 would be severe (>30 percent for Na). Sodium losses are significant only to the extent that Na aids in establishing osmotic potentials allowing cell expansion. Sodium is not a nutritional component of trees. The stands were marginal for clearcutting because of an apparent high level of removal of ecosystem copper, and in some cases Zn and Na. Stands 2 and 3 appear to be best able to withstand clearcutting because of low total ecosystem nutrient losses from bole removal. Stand 7 had high variability among subplots. Table 1 also lists the species composition and habitat type of each stand.

Table 1—Projected percent nutrient losses before treatment for seven stands at Lubrecht if the original stands were to be clearcut and boles only removed (to a 7-cm top). Spacing does not represent existing spacing in each stand. Treatments were not applied here. Nutrient losses should be the same within a stand for all projected treatments if stand density, tree size, and soils were uniform

Nutri-	Stand No. 1[1]				Stand No. 2				Stand No. 3				Stand No. 4				Stand No. 5			Stand No. 6			Stand No. 7		
ent	C	3	4	6	C	3	4	6	C	3	4	6	C	3	4	6	C	3	4	C	3	4	C	3	4
	Spacing (meters)[2]																								
Ca	5	5	4	5	6	3	6	6	3	3	3	2	6	4	4	6	5	13	14	5	5	6	14	9	12
Cu	40	33	34	39	24	19	26	28	27	20	23	24	27	20	24	21	27	35	37	15	18	16	35	52	29
Fe	5	7	5	5	6	7	9	9	5	4	6	5	5	4	5	5	3	3	3	3	3	4	7	6	5
K	7	7	8	7	8	4	8	8	8	8	9	8	10	11	12	14	14	14	15	14	16	17	16	12	14
Mg	8	7	7	7	8	4	8	8	7	7	6	5	11	8	9	12	18	19	21	13	16	17	14	12	15
Mn	8	11	9	8	15	8	13	10	9	12	12	9	15	20	12	17	28	25	31	22	24	28	33	23	30
Na	21	18	17	22	12	10	17	13	18	16	19	20	26	33	31	28	43	39	44	34	34	40	43	46	51
Zn	25	23	28	21	24	21	28	23	18	14	18	18	30	31	23	32	55	50	55	27	29	29	59	44	53

[1]Stand 1 = Gate of Many Locks, DF 55 percent, PP 40 percent, WL 5 percent, Shooflin soil, PSME/SYAL, CARU ht.[3].
Stand 2 = Shoe String - DF 60 percent, PP 30 percent, WL 10 percent, Shooflin soil, PSME/SYAL, CARU ht.
Stand 3 = Upper Sec. 16 - PP 90 percent, Greenough soil, PSME/SYAL, CARU ht.
Stand 4 = Baker Road - DF 50 percent, PP 40 percent, WL 10 percent, Crest soil, PSME/SYAL, CARU ht.
Stand 5 = Bottle Neck, PP - LPP 90 percent, Greenough soil, PSME/VACA ht.
Stand 6 = Coyote - WL 90 percent, Holloway or Beta soil, PSME/LIBO, VAGL ht.
Stand 7 = Sec. 12 Lower, LPP 90 percent, Greenough soil, PSME/VACA ht.
Total nitrogen is missing because it is not produced primarily from weathering.
[2]3 = 3 by 3 spacing; 4 = 4 by 4 spacing; 6 = 6 by 6 spacing.
[3]C = control, DF = Douglas-fir, PP = ponderosa pine, WL = western larch. See Pfister and others (1977) for habitat type abbreviations.

CONCLUSIONS

The NUTROSS model can be used economically to examine the impacts of various levels of harvest on ecosystem available nutrient losses. Under clearcutting, stands 1, 5, 6, and 7 would be subjected to nutrient losses in excess of 25 or 30 percent, the projected cutoff point for sound harvest practices. The model not only identified the chemically fragile soils, but it shows that Cu, Zn, and Na are the nutrients most likely to be adversely impacted by excessive harvest. The model also showed considerable variability in percent nutrient losses among treatments within a stand. These variations resulted from differences in original stocking level, species composition, or soil chemistry.

By September of 1990, the NUTROSS software will be available for distribution by the authors.

ACKNOWLEDGMENTS

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Speakers answered questions following their presentations. Following are the questions and answers on this topic:

Q.—N is the only element shown in regional nutrition studies to be limiting to tree growth; P, K, and S also may be limiting. How can you relate micronutrients (Zn, Cu, etc.) to tree growth? There does not seem to be that much demonstrated connection between tree growth and micronutrients.

A.—Early studies with western larch (Behan) and scotch pine have shown that trees require micronutrients in order to grow normally. The Australian literature has prolific references to deficiencies of Cu and Zn that have been demonstrated to reduce tree growth. (Carlyle and others 1989, Canadian Journal of Forest Research, and many others). Our research on xylem sap chemistry in western Montana has shown that nutrient imbalances, particularly trace metal deficiencies, do exist in geologically young soils or calcareous soils and that these are responsible for limiting tree growth (Stark and others 1989, Canadian Journal of Forest Research). Likewise, an accumulaton of trace metals in the understory vegetation has been demonstrated to correlate strongly with poor tree growth on geologically young soils such as are common in the mountainous regions of the Northwest (unpublished data). No forester wants to beleive that trace metals are important to tree growth, but the evidence is piling up rapidly. If you try to grow tree seedlings using pure chemicals with only macronutrients, they will not grow.

Q.—In many of our stands, only some of the branches are removed to a landing and burned (approximately 50 percent). Will NUTROSS handle this kind of scenerio?

A.—Yes, with a minor manipulation.

VALIDATION OF SOIL-SITE MODELS

David L. Verbyla

ABSTRACT

Hundreds of soil-site models have been published without being validated; such models may have prediction bias. The potential for prediction bias is especially high when many candidate predictor variables from a small sample are tested for during model development. Because of potential prediction bias, all soil-site models must be validated before being accepted. Two resampling procedures, cross-validation and the bootstrap, are introduced as simple statistical methods of validating soil-site models. These resampling methods provide a nearly unbiased estimate of the expected accuracy of a model. They are simple to computer program, and require no new data. The author recommends that soil scientists use a resampling procedure for the initial validation of soil-site models prior to expensive field validation.

INTRODUCTION

Forest site quality in the Rocky Mountains is often expressed as site index—the average height of dominant and codominant trees at a base age of 50 or 100 years. Site index must be indirectly estimated where site trees are unavailable for direct measurement. A common indirect method is the soil-site model where site index is modeled as a function of soil, topographic, and vegetation factors. This approach has been accepted since the 1950's, and hundreds of soil-site equations have been published (Carmean 1975; Grey 1983).

However, many of these soil-site models have been published without validating them. The objective of this paper is to demonstrate that soil-site models can have severe prediction bias and therefore must be validated as part of the modeling process. I will then introduce some simple statistical validation techniques that require no new data and provide a nearly unbiased estimate of model accuracy.

PREDICTION BIAS

Suppose we measure site index and soil pH from two forest stands. We can then develop a regression model that predicts site index as a linear function of soil pH (fig. 1). The model has a high apparent accuracy; the site index of the two stands is perfectly predicted by our regression model. However, the model probably has prediction bias because the actual accuracy of the model is probably less than perfect prediction.

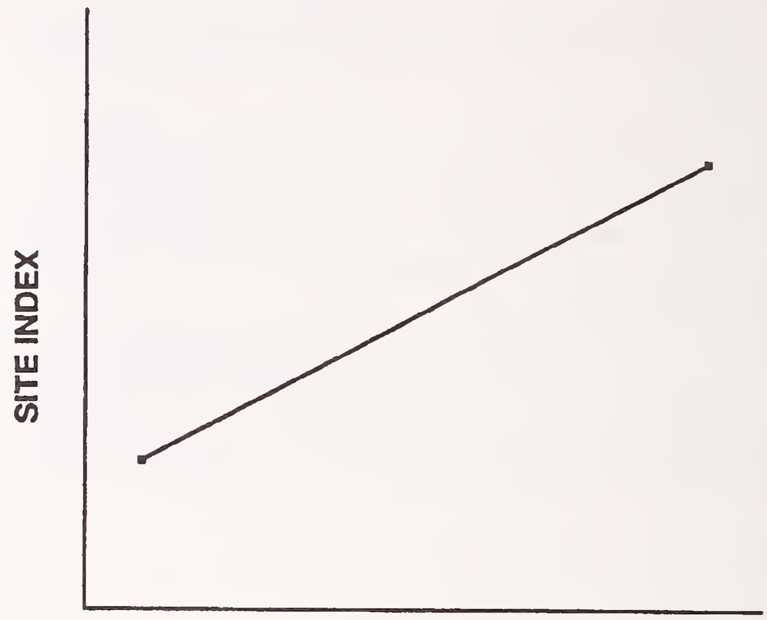

Figure 1—Linear regression based on two hypothetical sample cases.

The potential for prediction bias is great if many predictor variables are used in the model and the sample size is small. This is because spurious correlations (due to chance) may be incorporated in the model if many potential predictor variables are tested during model development. For example, I developed a regression model that had an R^2 of 0.99 and a linear discriminant model that correctly classified 95 percent of the sample cases; however, both these models were totally useless because they were developed with random numbers (Verbyla 1986). McQuilkin (1976) illustrated the same prediction bias problem by developing a soil-site regression with real data. His regression equation had an R^2 of 0.66; but when it was validated with independent data, the correlation between the actual and predicted site indices was less than 0.01 (McQuilkin 1976).

MODEL VALIDATION BY RESAMPLING METHODS

Because of potential prediction bias, soil-site models must be validated before being accepted. An intuitive approach is to randomly save half the sample cases for validation purposes. However, this is not a good idea. Consider figure 2: 20 sample cases are predicted by the linear discriminant boundary with an apparent accuracy of 90 percent. If we randomly select 10 sample cases to be excluded from model development (essentially sacrificed for model validation), two problems occur (fig. 3).

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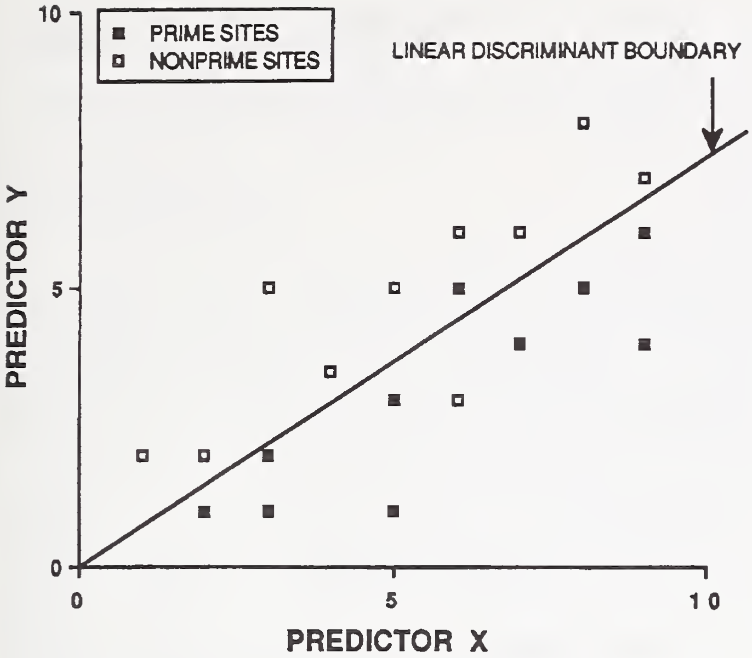

Figure 2—Linear discriminant boundary based on 20 hypothetical sample cases.

First, we do not have a reliable estimate of the slope of the linear discriminant boundary (also our model degrees of freedom are reduced by half). Second, we only have one validation estimate of model accuracy, and this estimate is not very precise (fig. 3).

Fortunately, there are better statistical procedures for validating models. One method, called cross-validation (or the jacknife) has been used in development of soil-site models (Frank and others 1984; Harding and others 1985). Cross-validation yields n validation estimates of model accuracy (where n is the total number of sample cases).

The cross-validation procedure is:

1. Exclude the i th (where i is initially one) sample case and reserve it for validation.
2. Develop the model with the remaining sample cases.
3. Estimate the model accuracy by testing it with the excluded sample case.
4. Return the excluded sample case, increment i, and repeat steps 1 through 4 until all sample cases have been used once for model testing.

The mean of the n estimates from step 3 is a nearly unbiased estimate of the expected accuracy of the model (if we were to validate it with new data from the same population) (Efron 1983).

A more precise estimate of expected model accuracy can be obtained using the bootstrap resampling procedure (Diaconis and Efron 1983; Efron 1983). The bootstrap resampling procedure is:

1. Randomly select "with replacement" n cases from the original sample. "With replacement" means that any sample case may be selected once, twice, several times, or not at all by this random selection process.
2. Develop the model with the selected sample cases.
3. Estimate the model accuracy by testing it with all sample cases that were not selected for model development in step 1.

The process is repeated a large number of times (200-1,000). The expected model accuracy is then estimated as the weighted mean of the estimates from step 3.

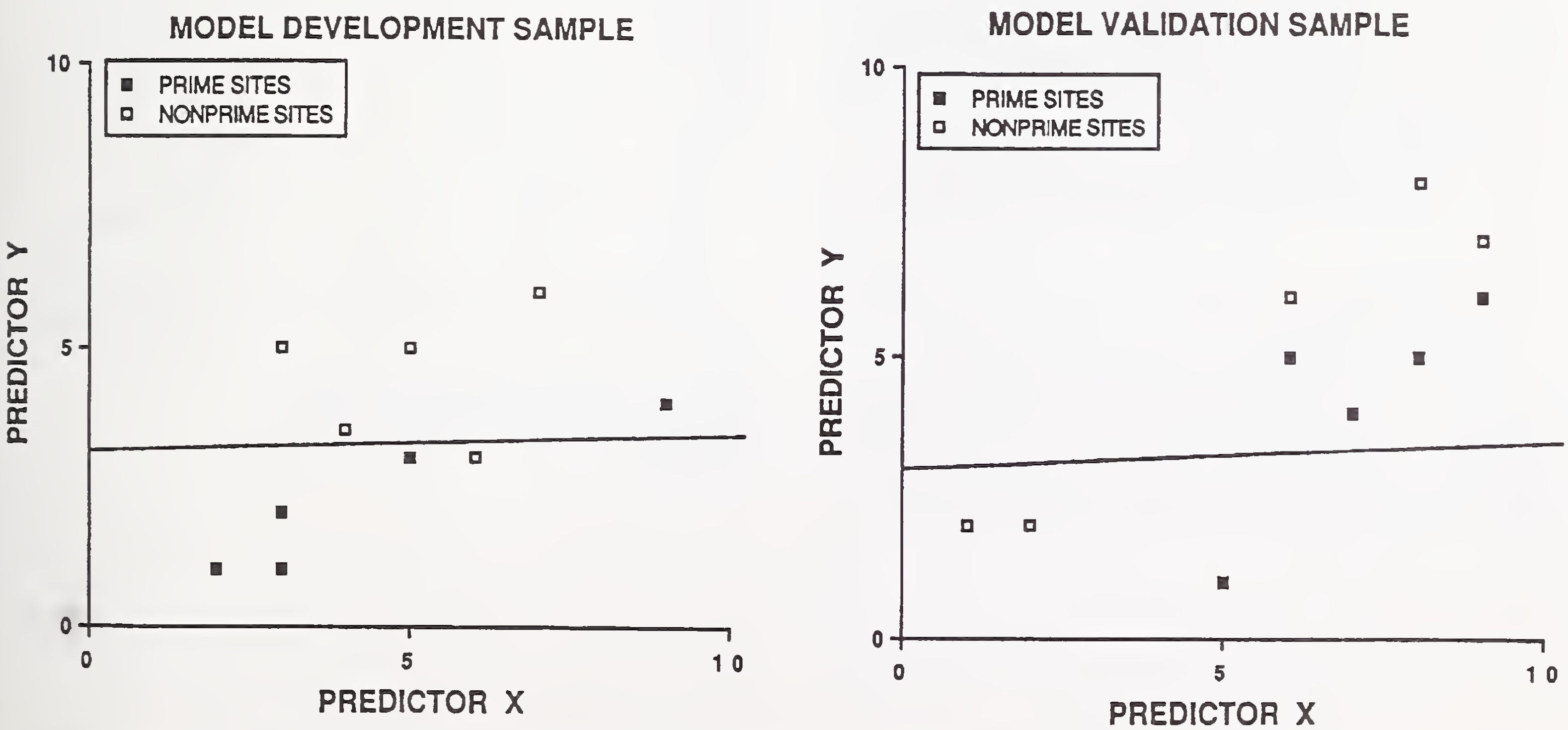

Figure 3—Random selection of half the original sample for model development and the remaining half for model validation.

COMPUTER SIMULATION

I will present computer simulation results to illustrate these methods. My example uses a model developed with discriminant analysis; however, these resampling methods can be applied to most predictive statistical models such as linear regression and logit models.

In this hypothetical example, we are interested in developing a model that predicts prime sites versus nonprime sites from soil factors. In the simulation, 30 sample cases (simulated forest stands) were generated with 10 predictor variables (simulated soil factors). The linear discriminant analysis procedure assumes normal distributions and equal variances, therefore the predictor variables were generated with these properties. Because each stand was randomly assigned to be either a prime site or nonprime site, the expected classification accuracy of the model was 50 percent (no better than flipping a coin).

The simulation was repeated 1,000 times. In reality, the modeling process is performed only once. If we use the original sample cases to develop the model and then test the model with the same data (called the resubstitution method), we would have a biased estimate of the model's accuracy. On average, the model would appear to have a classification accuracy of 75 percent (fig. 4). Yet, the actual accuracy of the model would be expected to be only 50 percent if it were applied to new data.

The same simulation was conducted using the cross-validation and bootstrap resampling methods to estimate model accuracy. Both methods produced nearly unbiased estimates of the expected accuracy of the model (fig. 5). The bootstrap method produced a more precise estimate and therefore is the best available method for estimating model accuracy (Efron 1983; Jain and others 1987).

CONCLUSIONS

Predictive statistical models can be biased. The prediction bias potential is especially high if sample sizes are small and many candidate predictor variables are tested for possible inclusion in the model. Because of the potential for prediction bias, predictive models must be validated. Resampling procedures such as cross-validation and the bootstrap require no new data and are relatively simple to implement (Verbyla 1989). There is no excuse not to use them.

A rational modeling approach is needed. The reliability and biological significance of predictive statistical models should be questioned (Rexstad 1988; Verbyla 1986). I believe that after models are developed, they should next be validated using a resampling procedure such as cross-validation or the bootstrap. The "acid test" should then be field validation to determine how well they predict under new conditions.

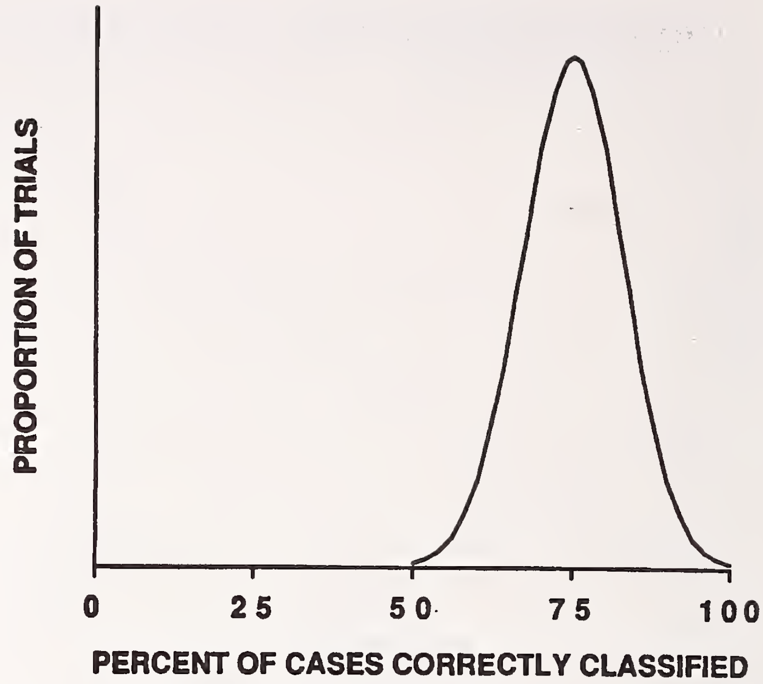

Figure 4—Smoothed frequency distribution (N = 1,000 simulation trials) of resubstitution method estimates of model classification accuracy.

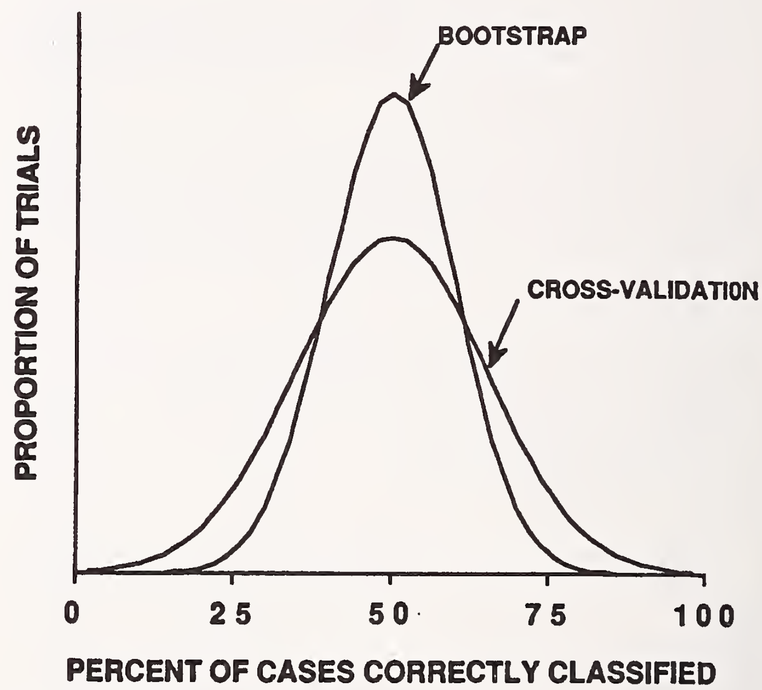

Figure 5—Smoothed frequency distribution (N = 1,000 simulation trials) of cross-validation and bootstrap estimates of model classification accuracy.

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ECOLOGICAL CLASSIFICATION AND CUMULATIVE SOIL EFFECTS

Mark E. Jensen

ABSTRACT

The Northern Region of the Forest Service, U.S. Department of Agriculture, has developed an efficient and consistent method for assessing cumulative effects of management practices on the soil and vegetation resources it manages. Ecological classification and analysis constitute the basis of this method. Since many of the management activities in the Region alter the present vegetation of a site (and consequently its values for a variety of resource uses) an understanding of plant succession relationships is critical to proper cumulative effects analysis. The ecological approach to cumulative effects analysis presented in this paper has proven effective in the Northern Region and is applicable to other wildland areas.

INTRODUCTION

The Forest Service has recently developed general guidelines (USDA FS 1988) for utilization of ecological classification and mapping in National Forest planning. This approach to land-use planning utilizes basic concepts of ecological classification (RISC 1983) in defining lands with similar potentials for management. National direction for a systematic approach to ecological analysis of cumulative effects has not been developed.

The primary objective of this paper is to discuss how ecological classification may be used in assessing the cumulative effects of management practices upon a variety of resources (for example, soil productivity, wildlife habitat, cattle forage, and watershed hydrologic function). A secondary objective is to describe some of the analysis software the Northern Region of the Forest Service uses in ecosystem cumulative effects analysis.

ECOLOGICAL CLASSIFICATION

Most ecological classifications utilize indicator plant species to describe environments with similar potentials for management. Habitat type classification based upon potential vegetation (associations) (Daubenmire 1952, 1968; Hironaka and others 1983; Jensen and others 1988; Pfister and others 1977) is an example of ecological classification that is widely used by various land management agency personnel, since relatively few diagnostic species are required to determine a site's ecological potential. In developing habitat type classification, minimally disturbed late-seral (potential natural community) or climax plant communities are sampled to determine which combinations of plant species indicate distinctive environments for management (Pfister and others 1977). Vegetation, soils, and other site information are collected at sampled plots to fully describe the environment indicated by a given habitat type.

Occasionally the range of environment that a habitat type occupies is sufficiently broad that it is necessary to further delineate it into a smaller classification unit to meet management needs. Such delineations are referred to as ecological sites by the Range Inventory Standardization Committee (RISC 1983), ecological types (site types) by the Forest Service (USDA FS 1988), or range sites by the Soil Conservation Service (Shiflet 1973). Ecological sites, ecological types, and range sites are similar in that each represents "a kind of land with a specific potential natural community (a habitat type) and specific physical site characteristics, differing from other kinds of land in its ability to produce vegetation and to respond to management" (RISC 1983).

Hierarchical levels of ecological classification may be developed to describe land potential for specific management needs. For example, in broad regional analysis, description of land potential to the formation level of vegetation classification (for example, grassland and forestland) may be adequate for planning purposes. In detailed project work site types, ecological types, ecological sites, or range sites are commonly used to describe the land's potentials for management. The hierarchical ecological classification levels used by Forest Service personnel of the Northern Region (Hann and others 1988) allow for flexibility in describing land potential dependent upon analysis scale and precision of interpretation needs (table 1).

PLANT SUCCESSIONAL CLASSIFICATIONS

Once ecological classification units are described for an analysis area successional plant communities are sampled to denote the various types of communities and successional pathways that may exist within a given ecological unit. Paired-plot sampling is commonly employed (Arno and others 1986), where treated stands of vegetation and adjacent untreated "controls" are sampled to facilitate accurate assessments of the ecological unit (on the control stand) and the successional plant community response of the treated stand. Vegetation, soil, site, and disturbance information (Hann and others 1988) are collected at each plot in this approach.

Community analysis (Keane and others 1988) of plot data is performed to combine plots into similar "existing vegetation" classification groupings. Ordination software (for example, DECORANA and TWINSPAN; Hill 1979a,b)

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Table 1—Description of hierarchical ecological classification levels utilized by the Forest Service Northern Region (Hann and others 1988)

Classification level	Example name	Appropriate analysis scale	Potential cattle forage
		(map scale)	*(lb/acre/yr/dry wt)*
Ecoregion	Northern Rockies	1:500,000	N/A
Geo-climate zone	Moist, ash-influenced mountains	1:250,000	N/A
Land form	Unstable uplands	1: 60,000	N/A
Formation	Forest lands	1: 60,000	0-2,000
Series	Grand fir forests	1: 60,000	50-1,500
Habitat type	Grand fir/ wild ginger	1: 24,000	200-1,000
Habitat type phase	Grand fir/ wild ginger/ yew	1: 24,000	300-800
Site type	Grand fir/ wild ginger/ yew—sandy substrate	1: 15,840	500-600

and similarity indices (Gauch 1982) are commonly used in such analyses. Dependent upon analysis objectives, existing vegetation classification groupings may be broadly (cover type) or narrowly (community type) defined (Hann and others 1988).

Prediction of plant community response following disturbance within ecological units may be facilitated by empirical or mechanistic succession models (Keane 1987). The application of "expert systems" technology to plant succession prediction is particularly useful in situations where limited data exist for the development of empirically based models (Keane and others 1988).

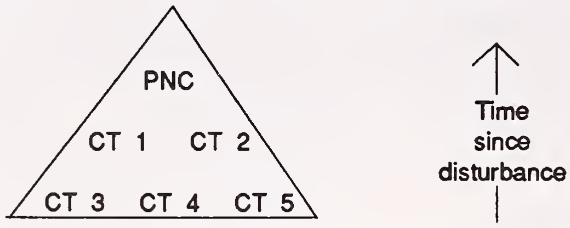
Once the existing vegetation types are described for an ecological unit, they are arrayed to display successional pathway relationships and correlated to type of disturbance. The "cone model" of plant succession (Huschle and Hironaka 1980) is a useful method for conceptualizing successional relationships within an ecological unit. This model assumes that, following disturbance, numerous "early seral" plant communities may develop on a site dependent on type of treatment, pretreatment vegetation composition, and gene pool access to the treated site. With increasing time after disturbance, species replacement occurs, which acts to narrow the range of communities that may exist on a given ecological unit. Given sufficient time (and absence of major disturbance), one plant community (potential vegetation) will be found on an ecological unit.

The cone model concept is illustrated in figure 1, which displays some generalized plant succession relationships within the grand fir/wild ginger (*Abies grandis/Asarum caudatum*) habitat type of northern Idaho (Green and Jensen 1989, this proceedings). Different types of disturbance contribute to multiple-plant successional pathways in this habitat type. Heavy soil displacement (removal of surface-soil ash cap) commonly results in the development of a forb-rich community (table 2) with few tree species present. This community type (CT 4) is persistent and does not experience significant species replacement with time. Low soil displacement in this habitat type initiates a successional sequence (CT 3 to CT 2 to CT 1 to PNC), which favors the establishment of tree and shrub species (fig. 1, table 2). Grass seeding (CT 5) delays the establishment of shrub species on low-soil-disturbance sites.

Documentation of plant successional pathways can only be developed within a reasonably defined ecological unit (Green and Jensen, this proceedings). Since the establishment of a given community type is a function of disturbance type and environmental variables (for example, climate and soil), it is critical that successional pathway predictions be developed within narrowly defined ecological units (the variability due to environment must be accounted for before disturbance relationships can be elucidated). The development of successional pathway predictions by ecological units provides a powerful tool for assessing the cumulative effects of management practices on vegetation, which in turn influences the land's value for multiple-use management.

VALUE RATINGS AND DESIRED COMMUNITY IDENTIFICATION

Each of the plant communities displayed in figure 1 possesses different values for a variety of resource uses (table 3). Such values are referred to as "Resource Value Ratings-RVR's" by RISC (1983) and are defined as "the



Disturbance	Time Since Disturbance (yrs)
	0 30 100 200 300
Clearcut, low soil displacement, grass seeding	CT5--------->CT5--------->CT2--------->CT1--------->PNC
Clearcut, low soil displacement	CT3--------->CT2--------->CT1--------->PNC
Clearcut, high soil displacement	CT4--------->CT4--------->?

Figure 1—Cone model representation of plant community type successional pathway development within the grand fir/wild ginger habitat type.

Table 2—Generalized vegetation description of plant community types found on the grand fir/wild ginger habitat type illustrated in figure 1. The numbers provided indicate percent foliar canopy cover

	Community type					
Dominant species	**PNC**	**CT1**	**CT2**	**CT3**	**CT4**	**CT5**
Grand fir (*Abies grandis*)	80	64	20	3	0	4
Douglas-fir (*Pseudotsuga menziesii*)	0	16	40	4	1	3
Twinflower (*Linnaea borealis*)	5	5	5	2	0	0
Blue huckleberry (*Vaccinium globulare*)	5	10	15	24	0	2
Thimbleberry (*Rhubus parviflora*)	0	0	2	2	5	7
Mountain maple (*Acer glabrum*)	2	5	20	5	0	0
Columbia brome (*Bromus vulgare*)	2	2	3	1	3	20
Bracted strawberry (*Fragaria vesca*)	0	0	0	3	30	4
Canadian thistle (*Cirsium arvense*)	0	0	0	3	10	0
Wild ginger (*Asarum caudatum*)	2	1	1	0	0	0

Table 3—Resource Value Ratings associated with several plant community types of the grand fir/wild ginger habitat type. The first number in each column represents the community types' absolute RVR value and the second number represents its relativized RVR value (percent of maximum in the column for the habitat type)

	Resource Value Ratings				
Community type	Tree basal Area	Elk hiding	Elk forage	Cattle forage	Basal vegetation and litter ground cover
	Ft²/acre	*Percent*	------ *Lb/acre/yr* ------		*Percent*
PNC	200 (80)	12 (20)	400 (52)	200 (33)	97 (100)
CT1	250 (100)	20 (33)	450 (59)	200 (33)	96 (99)
CT2	150 (60)	60 (100)	600 (78)	220 (37)	95 (98)
CT3	10 (4)	10 (17)	700 (91)	400 (67)	85 (88)
CT4	2 (1)	0 (0)	650 (84)	380 (63)	70 (72)
CT5	8 (3)	1 (2)	770 (100)	600 (100)	80 (82)

value of vegetation present on an ecological site for a particular use or benefit." RISC further states that "RVR's may be established for each plant community capable of being produced on an ecological site, including exotic or cultivated species."

Displaying RVR's by plant community groupings allows the user to decide which plant community best meets management objectives for a given analysis area. Such plant communities are referred to as the "Desired Plant Community" of the ecological unit (USDA FS 1988) and may be used to rate the floristic similarity of other community types (ecological status) to the target community in an analysis area. Low similarity measurements would indicate a need for management action; a high similarity measurement would indicate little or no need for management action in this approach. The RVR's associated with the existing plant communities of the grand fir/wild ginger habitat type (table 3) are useful in illustrating these points.

The grand fir/wild ginger habitat type is an important component of managed forestlands of northern Idaho (Green and Jensen 1989). Timber harvesting is common in this habitat type; however, elk habitat, livestock grazing, and watershed hydrologic function are also important issues in multiple-use management in this type. The desired plant community of this habitat type should provide optimum timber, wildlife, range, and watershed resource values.

The forb-rich community type (CT 4), which is promoted by high soil displacement, is clearly not the desired plant community of this type, since it has the lowest relativized resource values (table 3) for elk hiding cover (0 percent), tree basal area (1 percent), and ground cover (72 percent). In this example CT 2 is the desired plant community, since it has the optimum combination of resource values given the management issues of concern. Management practices that promote the development of this community type include clearcutting with low soil displacement. Accordingly, timber harvesting in this habitat type should avoid heavy soil displacement and grass seeding. This will promote rapid development of the desired plant community.

This example is simplified. In practice, more than one desired plant community expression of an ecological unit may be required to meet management objectives for an area. For example, 60 percent of an analysis area may be targeted for maximum timber production and 40 percent for elk habitat emphasis. In this situation, 60 percent of the area would have the CT 1 community type as the desired plant community for timber objectives, and 40 percent of the area would have CT 2 as the desired plant community for elk management objectives (table 3). Management practices would then be scheduled that promoted establishment of the desired plant communities (for example, clearcutting or thinning of PNC stands with minimal soil displacement) and trend monitoring would be conducted over time to ensure that desired plant succession pathways were being followed. The spatial distribution of the desired plant communities in an analysis area is important to many wildlife species (for example, those needing migration routes) and must be considered in the planning process. Digitization of existing vegetation and ecological unit maps for geographic information systems analysis is extremely useful in addressing spatial questions related to desired plant community distribution.

FOREST SERVICE APPLICATION EXAMPLE

The Northern Region of the Forest Service utilizes ecological classification in describing the cumulative effects of management practices on a variety of ecosystems (for example, grasslands, forestlands, and riparian areas). Multiple-use management of these lands requires that the effects of management activities be documented in a consistent, efficient manner. The first step in meeting this task is to ensure that different resource functions (for example, wildlife, timber, and soils) utilize common terms and databases when characterizing and analyzing the ecosystems they manage. Standards for ecosystem characterization (for example, soils, vegetation, and climate) have been developed for use throughout the Region by the Ecosystem Management Group (Hann and others 1988) and the Timber Management Group (USDA FS 1989b, 1986c). Data analysis and prediction systems (Keane and others 1988) have also been developed to ensure consistent interpretation of ecosystem data.

Some of the various databases and analysis programs (Keane and others 1988) utilized by Northern Region personnel in cumulative effects analysis are displayed in figure 2. Two basic types of polygon map level information are utilized in cumulative effects analysis in the Region. One of these polygons is land potential (ecological unit), which is delineated by soil survey map units (SOILMUD). Attribute data linked to these polygons include soils, geology, climate, and ecological classification. The second polygon represents existing vegetation, which is documented through vegetation stand mapping units (VEGSTAND). Vegetation stands consist of vegetation based polygon delineations that have unique automated data processing (ADP) code identifiers (for example, timber stands). Management activities and plot-level information are linked to these polygons for data analysis. The combination of these two polygons through relational databases and geographic information systems software allows the user to identify site potential, successional pathways, and resource values of an analysis area in an efficient manner.

Classification databases are constructed from plot data and published information and are used to characterize

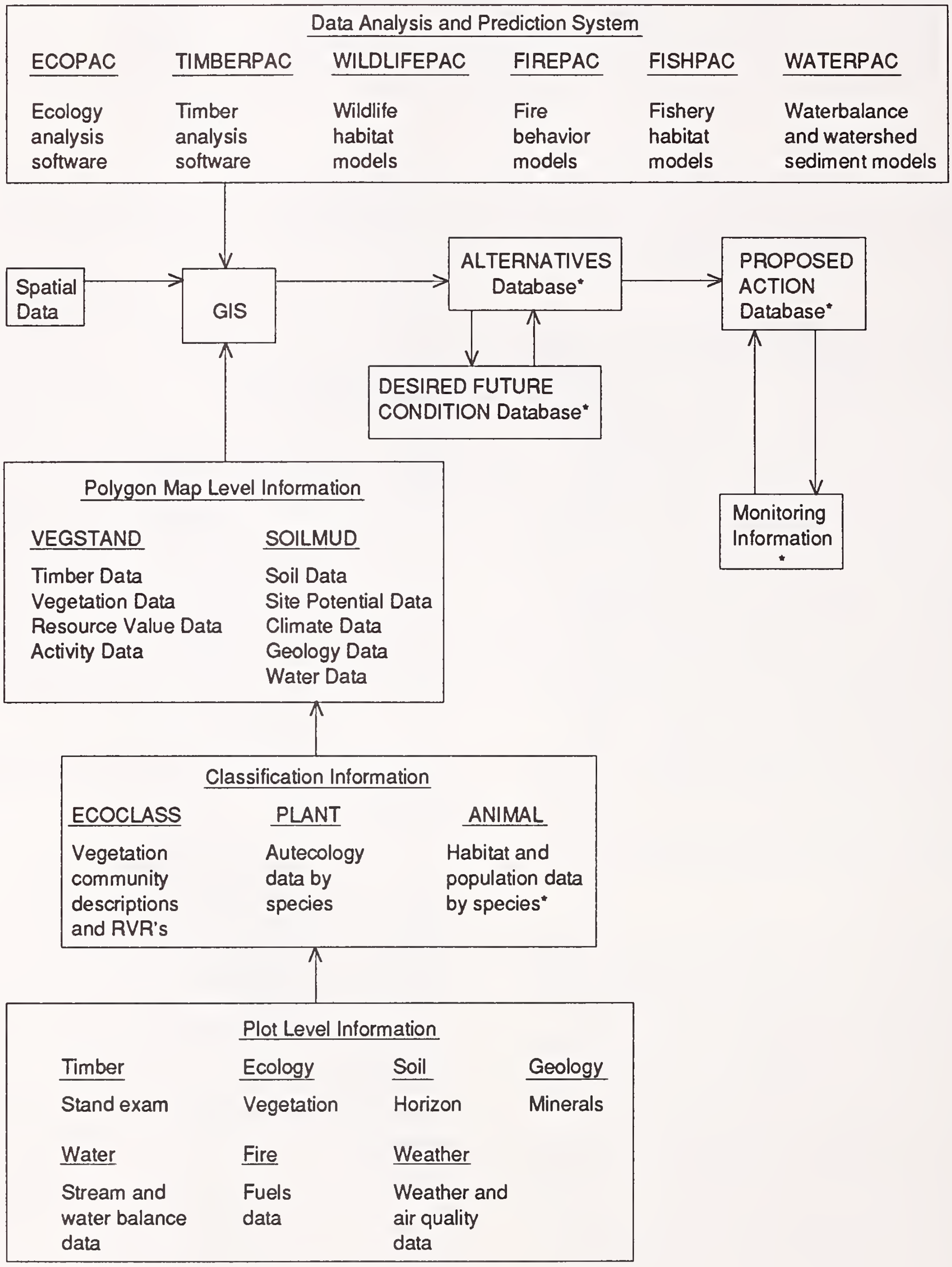


Figure 2—Listing of some of the databases and analysis programs used by the Forest Service Northern Region in cumulative effects analysis. * indicates databases that are being developed.

both the vegetation stand and soil survey map units. The ECOCLASS database (fig. 2) contains information on resource value ratings for seral plant communities; the PLANT database contains information on plant species autecological relationships, which is used to predict plant succession through "expert systems" technology; and the ANIMAL database contains information concerning wildlife species habitat requirements and population dynamics.

Numerous data analysis and prediction systems are linked to the polygon map level databases to facilitate interpretation of management effects on various resources (for example, fire behavior, watershed hydrologic function, and wildlife habitat suitability). Outputs from such analyses are stored in an ALTERNATIVES database, which allows the user to document the types of resource response associated with different management activities on a given map unit. Such output is contrasted to the DESIRED FUTURE CONDITION database which displays the desired characteristics of a map unit given management objectives for an area. Selected alternatives for management are documented by map polygon in the PROPOSED ACTION database. Monitoring information is linked to the PROPOSED ACTION database to ensure that proposed activities are implemented and that data analysis and prediction systems output for the proposed action were reasonable. The process used by Northern Region personnel in applying ecological classification concepts to cumulative effects analysis is outlined by Jensen and others (1991).

CONCLUSIONS

Utilization of ecological classification concepts in cumulative effects analysis provides an improved method for development of documents required by the National Environmental Policy Act. Since many management activities on Federal land alter vegetation, which in turn influences the land's value for a variety of resource uses, it is important that reasonable predictions of plant community successional response be made prior to scheduling management actions. The approach to cumulative effects analysis presented in this paper has proven useful in describing the effects of management activities on lands managed by the Forest Service.

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DOES FIRE EXCLUSION INCREASE PRODUCTIVITY OF PONDEROSA PINE?

P. H. Cochran
W. E. Hopkins

ABSTRACT

Railroad logging in the 1920's and 1930's removed most of the old-growth ponderosa pine (Pinus ponderosa *Dougl. ex Laws.) from about 150,000 acres in central Oregon. Although the old-growth stands had frequent, light ground fires, most of the second-growth stands have not been subjected to fire. The soils (Xeric Vitricryands, developing on Mazama pumice and ash) now have a litter layer, and today, abundant shrubs, herbs, and residue occur in contrast to the historic open, parklike conditions. Prescribed underburning has reduced growth of basal area, height, and volume of second-growth ponderosa pine on these soils. Yield tables revised in 1961 were assembled from samples taken shortly after the start of fire protection. Predictions of gross periodic annual volume increments (PAI) from the 1961 data may be lower than actual PAI's for current stands that have no fire in their history, even when no change in site index is assumed.*

INTRODUCTION

Today, ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) forests on the eastern side of the Cascade Range and Sierra Nevada in Washington, Oregon, and California differ markedly from the forests that existed at the start of the 20th century. Dramatic ecological changes reflected in stand structure are apparent throughout the area (table 1). Stand structure as first encountered by settlers has been well documented (Langille and others 1903; Lieberg 1899; Munger 1914, 1917). Munger (1917), for example, described the large tracts of pure yellow pine forests in Oregon where "the trees are spaced widely, the ground is fairly free from underbrush and debris, and travel through (the forest) on foot or horseback is interrupted only by occasional pockets of fallen trees" (fig. 1).

The importance of fire in maintaining these open, park-like, seemingly even-aged stands was generally recognized. Jepsen (1923) states that the main silvicultural features of the Sierran forest, "that is, density, reproductive power, and dominance of types, are in great part expressions of the periodic fire status." Bork (1985) found that fires in the ponderosa pine forests in central Oregon occurred every 8 to 12 years before the beginning of fire protection and suppression in about 1917.

Ponderosa pine forests of today have abundant shrubs and herbs in the understory; thickets of pine regeneration in uneven patterns; down, decaying woody material; and a litter layer is usually present. Over the 70 years that these changes have taken place in the stands and their associated understories, the additions, subtractions, transformations, and translocations associated with nutrient cycling and soil genesis also must have changed considerably. Even now these processes are not likely to have reached equilibrium.

As the understory and soil surface have been modified by fire protection, the overstory of most of the ponderosa pine stands has been modified by harvesting. Cutting methods have ranged from light selection to clearcutting. The undisturbed ponderosa pine forest generally seemed to be even-aged, but, in reality, consisted of clusters of small even-aged groups. Forests today are even-aged after clearcuts or uneven-aged with two or more broad size classes. Few stands have an all-aged or all-sized distribution of trees. Many ponderosa pine stands have been precommercially thinned in the last three decades. Some of the resulting slash has been burned, but most has been either crushed or left untreated, adding to the residue on the forest floor.

Soil productivity can be thought of as the ability of the soil to supply water and nutrients necessary to produce plant material. Many of the soils in central Oregon are Xeric Vitricryands, developing on Mazama pumice and ash. These soils have low bulk densities, are cold, and also are low in nitrogen (N), phosphorus (P), and sulfur (S). When we noticed that old-growth stumps often had narrower ring widths than adjacent second-growth stumps, we began to consider the question of whether the productivity of these soils has changed with time since fire exclusion.

METHODS OF STUDY

To begin to answer the question, we measured growth rates of three second-growth stands on low, medium, and high sites. We then compared the estimated gross periodic annual cubic volume increments (PAI's) of these second-growth stands with predictions obtained from Meyer's (1961) yield tables with adjustments for density. Meyer's yield tables were assembled from samples taken shortly after the start of fire protection and should partially reflect the influence of fire on productivity if productivity is influenced by more than site index values. We further examined how gross PAI's for second growth would be reduced when values for site index were reduced in response to periodic underburning.

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Table 1—Fire climax ponderosa pine forest (1890-1920) compared to the climatic climax ponderosa forest of 1990

Characteristic	1900's	1990's
Trees per acre	10-30	100's-1,000's
Mean diameter—inches	16-18	<8-10
Dominant diameter—inches	25-34	25-34
Average height of dominant trees—feet	123	115
Average age—years	225-275	<150
Trees per acre (seedlings, saplings, poles 2-10 inches d.b.h.)	0-2	>100
Snags per acre	0-2	0-4
Shrub cover—percent	0-5	30-70
Herbaceous cover—percent	0-8	10-30

Figure 1—A ponderosa pine stand about 14 miles south of Bend, OR, in 1917.

Growth rates were determined for three second-growth, even-aged stands in the Deschutes National Forest for the 1979-83 and 1984-88 growing seasons. These stands originated after the old growth was clearcut during the railroad logging operations in the 1920's. All three stands had been precommercially thinned before 1963. The low site stand is in the Fort Rock Ranger District 14 miles southeast of Bend, OR. The medium site stand is about 14 miles south of Bend near the Sugar Cast timber sale area. The high site stand is in the Bend Ranger District about 9 miles west of Bend.

Nineteen square, 0.4-acre plots were established in each area. Each tree on each plot was tagged, and its diameter at breast height (d.b.h.) was measured with a caliper oriented in a north-south direction. Two increment cores were taken from each tree at 4.5 feet (b.h.), one core on the north and another core on the south side of the tree, from which radial wood growth for the 1979-83 and the 1984-88 periods was determined. The calipered outside bark d.b.h. and the radial wood growth measurements were used with an equation for bark thickness (DeMars and Barrett 1987) to determine the diameters and basal areas both inside and outside bark at the end of the 1978, 1983, and 1988 growing seasons.

A subsample of heights was measured with clinometers in fall 1988. For each plot, trees were divided into 2-inch diameter classes, and 15 trees for each plot were selected for height measurement. Trees were randomly selected within each diameter class so that the whole range of diameter classes was sampled about equally for heights. These heights (H) and their corresponding diameters were then used to determine the coefficients for the model

$$\log_e H = a + b/\text{d.b.h.}$$

for each plot. If the R^2 for this relation was less than 0.85 for any plot, five additional trees were randomly selected from the diameter class distributions for additional height determinations, and the model was refitted for the plot. These equations with separate coefficients for each plot were then used to determine heights for each tree in fall 1988.

To estimate heights in fall 1978 and 1983, five trees on each plot were felled. These trees were randomly chosen within the diameter class distributions so that the largest, the smallest, and three intermediate-sized diameter class distributions were sampled. After felling, whorls were counted back at 5-year intervals, and the stem was cut and a ring count was made so the height in 1978 and 1983 could be accurately determined. Equations of past heights (H_i) as a function of current height were fitted for each area with combined data from all 19 plots using the model $H_i = a_i + b_i H$, where i refers to 1978 or 1983. The individual a and b values for each area and year were then used to determine heights for each tree in 1978 and 1983.

Cubic-foot volumes (V) for each tree in 1978, 1983, and 1988 were then determined by using (DeMars and Barrett 1987)

$$V_i = 0.005454(D_i)^2(H_i)(F_i)$$

where D is the diameter inside bark, F is a cylindrical form factor that is a function of D and H, and i references the year 1978, 1983, or 1988.

Plot volumes were then determined from individual tree volumes for each year. From these plot volumes, PAI's were determined for the 1978-83 and 1984-88 periods. There was no evidence that mortality occurred on these plots during these periods so these increments are gross PAI's.

Site index values (Barrett 1978) were determined for each plot. Age at 4.5 feet and stand density index (SDI) also were determined for the midpoint of each period for

each plot. The relation of trees per acre (T/A) to quadratic mean diameter (Dg) in inches for "normal" or "fully stocked" ponderosa pine derived from Meyer's original plot data is (DeMars and Barrett 1987)

$$\log_e(T/A) = 9.9658 - 1.7653 \log_e(Dg) \cdot \quad (1)$$

Therefore,

$$SDI = (T/A)(Dg/10)^{1.76563} \cdot \quad (2)$$

Estimates of gross PAI's for fully stocked stands (SDI = 365) were obtained by using the normal yield data in tables 3 and 6 in Meyer (1961). For this estimate, average volumes per tree were calculated for each decade. Next, we assumed that the average volume of the trees that died during the decade was equal to the average volume of all the trees at the beginning of the decade. The volume of mortality during the decade was calculated by multiplying the average volume for all trees at the start of the decade by the difference in the number of trees at the start and end of the decade. Gross PAI for the decade was then calculated by adding the mortality to the difference between live tree volume at the end and beginning of the decade and dividing the result by 10. The site index values of Meyer (SI) were converted to the site index values of Barrett (S) by using the relation

$$S = 37.735 + 0.931451(SI) \quad (3)$$

determined from the original plot data of Meyer (DeMars 1988). Total age (A_t) used by Meyer was converted to age at breast height (A) by using the equation

$$A = A_t - 6 - 0.0901(150 - SI) \quad (4)$$

derived from Meyer's (1961) definition of stand age and his example of age determination. These estimated gross periodic annual volume increments for fully stocked stands, gross PAI_m, were then related to Barrett's (1978) site index, S, and age at breast height, A, at the midpoint of each decade by using the model:

$$\log_e(\text{gross } PAI_m) = b_0 + b_1A + b_2S + b_3\log_eA + b_4\log_eS + b_5(S)(A) \cdot$$

To estimate gross PAI's of understocked stands using this relation, a density adjustment is necessary. Density adjustments were obtained from functions given by Barrett (1982) that relate gross PAI's for understocked stands with three different understory conditions (shrubs and grass, grass, no understory) to SDI. These functions were used to calculate gross PAI's for understocked stands as fractions of gross PAI's produced at full stocking. These increment fractions (gross PAI's of understocked stands/gross PAI's at full stocking) were then related to the fractions of full stocking (actual SDI/365).

Gross PAI's, obtained by using the relation obtained from yield table data with a density adjustment for a shrub understory, were predicted for both the 1979-83 and 1984-88 periods for each of the 57 sample plots in the second-growth stands. The actual gross PAI's were then compared with these predicted gross PAI's. These stands had not experienced any wild or prescribed fires. To further investigate the change in productivity due to fire exclusion, the gross PAI's for these plots were again predicted after adjusting the site index value downward for each plot by using the information from Landsberg and others (1984) and other information on file at the Silviculture Laboratory in Bend, OR. According to these data, height growth is reduced to 92 percent of the controls during the first 4 years after a moderate underburning. Some recovery in height growth occurs during the fifth through eighth years after moderate underburning, with perhaps a full recovery during the ninth through twelfth years after the initial underburn in these kinds of stands on these soils. By using Bork's (1985) most conservative fire frequency of one fire every 12 years for central Oregon, we estimate that site index values would be 96 percent of the value for the same plot if the plot was not burned. The actual gross PAI's were compared with the predicted gross PAI's determined by using the reduced site index estimates.

RESULTS

The second-growth stands on the three sites in fall 1988 had fairly similar mean diameters (table 2). Average density and volume increased with increasing average site index, and average age decreased with increasing site index.

Gross periodic annual cubic volume increments for fully stocked stands estimated from Meyer's (1961) yield tables (gross PAI_m) are described by (fig. 2):

$$\log_e(\text{gross } PAI_m) = 4.2771 + 0.01995(S) - 0.01233(A) - 0.60375(\log_eS) + 0.46965(\log_eA) + 0.000012829(S)(A) \cdot \quad (5)$$

Density adjustments from Barrett (1982) describe the gross PAI of an understocked stand as a fraction of the

Table 2—Average values for 19 0.4-acre plots in each of three second-growth stands in fall 1988

Characteristic	Fort Rock	Sugar Cast	Bend District
Barrett's site index—feet	82.0	102.0	115.0
Age at b.h.—years	56.0	49.0	40.0
Mean diameter—inches	9.9	9.5	9.3
Average height—feet	43.1	50.7	43.6
Trees per acre	195.0	286.0	317.0
Stand density index	190.2	260.3	262.1
Basal area—ft²/acre	103.6	140.4	143.0
Volume—ft³/acre	1,613.2	2,797.6	2,562.2

gross PAI for a fully stocked stand (SDI equals 365) at the same age on the same site. These increment fractions, Y, as a function of the actual stand density index divided by 365, X, are (fig. 3):

$$\log_e Y = 0.58354 - 0.59704(X) + 0.66199(\log_e X) \qquad (6)$$

for stands without understory vegetation;

$$\log_e Y = 0.60118 - 0.61568(X) + 0.65261(\log_e X) \qquad (7)$$

for stands with an understory of grass; and

$$\log_e Y = 0.34724 - 0.3307(X) + 0.68687(\log_e X) \qquad (8)$$

for stands having an understory of grass and shrubs.

All 57 sample plots have an understory of grass and shrubs. The gross PAI's for these plots therefore were predicted by using equations (5) and (8). Differences between actual and predicted gross PAI's varied between periods and locations. Actual gross PAI's averaged 8.3 percent higher than the predicted gross PAI's with a 95-percent confidence interval of 6 to 10.5 percent (fig. 4, table 3). When the site index values were lowered by 4 percent in determining predicted gross PAI's to approximate the site index values before the advent of fire protection and suppression, actual gross PAI's averaged 14.6 percent higher than predicted gross PAI's. The 95-percent confidence interval in this case was 12.3 to 17 percent. The percent differences here were calculated by subtracting the predicted values from the actual values, dividing this difference by the predicted values, and multiplying the result by 100. The linear regression relating actual gross PAI's to predicted gross PAI's obtained using a reduced site index value has a slope of 1.19 and an intercept of –3.38. The R^2 value is 0.85, and the standard error of the estimate is 11.8 cubic feet per acre per year. The residuals for the equation relating actual gross PAI's to predicted gross PAI's (fig. 4) seemed to be balanced when plotted against age, site index, and SDI. The percent differences between actual and predicted gross PAI's were weakly correlated with age, SDI, or site index. Separate linear regressions of the percent differences where site index was not reduced as functions of age, SDI, and site index had R^2 values of 0.06, 0.04, and 0.01, respectively. Plots of the data did not indicate any nonlinear relations.

DISCUSSION AND CONCLUSIONS

The weak correlations of the differences between actual and predicted PAI's with site index, age, and SDI indicate that the model used reasonably described the general pattern of change in gross PAI over the ranges of these variables encountered in this study. The model may predict PAI's that are low, but growth-influencing factors like precipitation can vary greatly between different 5-year

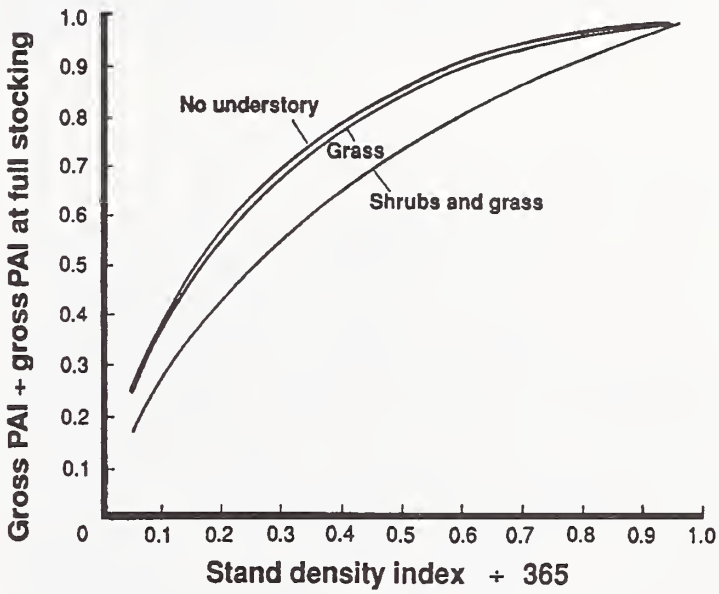

Figure 3—Fractions of the gross periodic annual cubic volume increments at full stocking produced on the same site at the same age at lower stand densities. The lower stand densities are expressed as fractions of the stand density index at full stocking, 365.

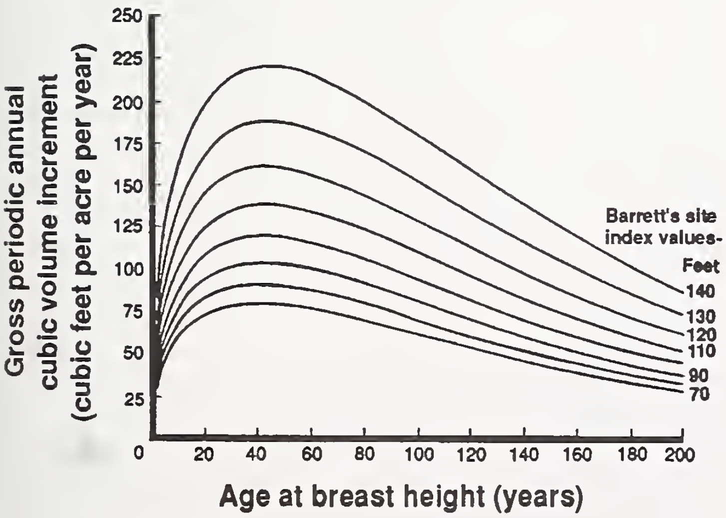

Figure 2—Gross periodic annual cubic volume increments for fully stocked, even-aged ponderosa pine stands estimated from Meyer (1961).

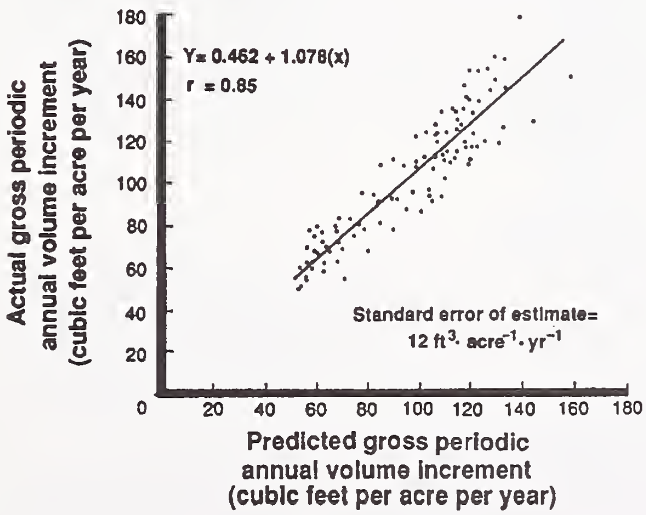

Figure 4—Actual versus predicted gross periodic annual cubic volume increments for the 1979-83 and 1984-88 growing seasons for 57 plots in second-growth ponderosa pine.

Table 3—Average percent differences and their confidence intervals between predicted and actual gross cubic volume PAI's for 19 plots in each of three unburned second-growth stands over two different 5-year periods[1]

Area	Fort Rock		Sugar Cast		Bend District		Combined areas,
	1984-88	1979-83	1984-88	1979-83	1984-88	1979-83	all periods
Percent difference assuming no change in site index	14.5	1.3	2.8	5.7	15.1	10.3	8.3
95 percent confidence interval	9.6, 19.5	−2.5, 5.0	−2.4, 8.1	−1.3, 12.7	9.6, 20.5	5.2, 15.5	6, 10.5
Percent difference assuming fire reduces site index[2]	19.6	5.7	9.1	12.0	23.2	18.1	14.6
95 percent confidence interval	14.5, 24.6	1.9, 9.5	3.8, 14.4	4.8, 19.3	17.5, 29	12.8, 23.5	12.2, 17

[1]Percent differences were calculated by using actual (a) and predicted values (p) in 100(a-p)/p.
[2]These percent differences assume underburning every 12 years would lower the site index values by 4 percent.

periods and between close locations in any 5-year period over the ponderosa pine type.

In this case study, fire history is confounded with other factors that may cause differing growth rates. These results therefore are not conclusive. The actual growth rates, however, were considerably higher than the growth rates predicted using current site index values for the 1984-88 period for the low site Fort Rock stand and for both periods for the high site stand in the Bend Ranger District. Further, actual growth rates for all areas and periods were higher than values predicted using lowered site index values. We do know from other studies in second-growth stands on these soils that underburning reduces basal area and height growth rates of trees that survive for at least 4 to 8 years after burning and possibly for longer. We feel that a reduction of site index by 4 percent due to repeated, moderate, underburning every 12 years on these soils is a conservative estimate.

Underburning to reduce fuel loads, to improve wildlife habitat, and to produce ponderosa pine stands with the characteristics of stands first viewed by the early settlers will continue to be prescribed. Continued long-term investigations of the influence of prescribed burning on productivity therefore, are necessary. These investigations should include the study of growth rates after fall and spring burns along with the processes that may be influenced by burning.

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Poster Papers

SOIL MANAGEMENT OF SEED ORCHARDS AND EARLY SELECTION TRIAL PLANTATIONS IN NORTHERN IDAHO

Gary L. Ford and Aram Eramian

Management of tree improvement areas is one of the most intensive uses of the soil resource in forestry today. Very site-specific soils information is required for the management of these areas. Practices used include extensive clearing, disking, ripping, irrigation, fertilization, and use of cover crops. A complete soil management program is essential to the success of tree improvement programs. This paper discusses soil management activities at two tree improvement areas in the Sandpoint Ranger District of the Idaho Panhandle National Forests.

TREE IMPROVEMENT PROGRAM

Tree improvement is a program that combines genetics and tree breeding to improve yields and overall quality of forest tree species. The goal of the Forest Service Northern Region Tree Improvement Program is to increase yields of wood on appropriate lands through genetics and tree breeding.

The Dry Creek Tree Improvement Area, located near Clark Fork, ID, is 80 acres in size and currently supports programs for Douglas-fir (*Pseudotsuga menziesii*) and western larch (*Larix occidentalis*). There are three early selection trial (EST) plantations and one established seed orchard on site. The purpose of the early selection trial plantations is to obtain early growth and hardiness data from a large number of families that have been selected from wild stands. The best individuals are selected and planted in seed orchards to produce seed for reforestation programs in that species' breeding zone.

When fully developed, Dry Creek will have two Douglas-fir seed orchards and three lodgepole pine (*Pinus contorta*) seed orchards producing genetically improved seed for reforestation programs in northern Idaho and western Montana. These five orchards will cover 50 acres and include 5,400 genetically improved trees.

The Grouse Creek Tree Improvement Area, located 18 miles northeast of Sandpoint, ID, will support seed orchards for western larch, western white pine (*Pinus monticola*), lodgepole pine, Englemann spruce (*Picea engelmannii*), and grand fir (*Abies grandis*). The site currently has an established 12-acre western white pine seed orchard. Approximately 120 acres of seed orchards will be established at Grouse Creek when it is fully developed in the year 2000.

SOIL MANAGEMENT OBJECTIVES

Soil management objectives within the two tree improvement areas are to: (1) locate seed orchards and early selection trial plantations on soils that are suitable for these uses, (2) manage soil fertility and soil moisture so tree improvement program objectives are met, and (3) protect long-term productivity.

SITE SELECTION PROCESS

A regional team first identified site factors considered important to development of a tree improvement area. These were: soils, climate, potential for disease and insects, quality and quantity of irrigation water, availability of power, and proximity to an all-season road.

The initial soil factors considered important were drainage, compactibility, fertility, and pH. The climatic factors identified were length of growing season, amount and pattern of annual precipitation, average temperatures and relative humidities for growing season, cold air drainage, and frost hazard.

SOIL MAPPING

An Order 2 soil survey was completed at the Dry Creek site in the fall of 1983 to determine if the site was suitable for use as a tree improvement area. In 1989, an Order 1 survey was conducted of the soils in each plantation and orchard site. Model soils were identified and sampled for laboratory analysis. Soil boundaries and modal soil locations were recorded on the orchard site map for future reference. Soil moisture monitoring stations have been installed on each modal soil close to where it was described and sampled.

An Order 2 soil survey was conducted at the Grouse Creek site in 1986 to determine if it was suitable for use as a tree improvement area. In the spring of 1990, potential sites for a western larch seed orchard were evaluated.

SOILS AND SOIL MANAGEMENT CONCERNS

The soils at the Dry Creek site have formed in volcanic ash over a variety of water-deposited parent materials. At Grouse Creek, most soils have formed in volcanic ash over glacial till. The principal management concerns for both areas are fertility, irrigation scheduling for the different soils, compaction from the clearing, and some minor areas with shallow, dense layers that restrict root growth and water movement.

CLEARING AND SITE DEVELOPMENT

Dry Creek was selected as a tree improvement site in 1983. Approximately 30 acres were first cleared to prepare sites for Douglas-fir and western larch early selection trial plantations. In August of 1987, an additional 40 acres of land was cleared to prepare sites for two lodgepole pine

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plantations. The Grouse Creek site was first cleared in 1986 to prepare sites for a western white pine seed orchard. Additional acreage was cleared in 1987-88 for a western larch seed orchard.

COMPACTION/RIPPING

In 1987, infiltration tests were conducted at the Grouse Creek site. Infiltration rates were found to be extremely variable and greatly reduced in some areas. Tests were conducted comparing bulk densities and infiltration rates. In general, the greater the compaction (bulk density), the lower the infiltration rate was found to be.

Random transects were established and bulk density samples were collected to determine the amount of soil compaction. These tests indicated that the site had been extensively compacted during the clearing process and the disk that had been used afterwards had only loosened the top 4 inches of soil. Research has shown that this compaction can reduce tree growth (Clayton and others 1987; Froehlich and McNabb 1984; Wert and Thomas 1981).

In the fall of 1988, soils at both the Dry Creek and Grouse Creek sites were ripped with a forest cultivator to reduce the compaction resulting from the clearing process. The compaction usually extended to a depth of approximately 12 inches, and the cultivator satisfactorily loosened the soil to that depth. The purpose of the ripping was to improve water infiltration and reduce bulk densities so root egress would not be restricted.

The cost of ripping the compacted areas at the tree improvement sites was $78.00 per acre for the first two passes with the forest cultivator. Additional passes, if required, cost an additional $36.00 per pass. Production rates were governed by the slope and amount of vegetation on the site. Average production rates experienced were 5 to 6 acres per day.

SOIL SAMPLING AND TESTING

Soil samples were collected from the Dry Creek site in May 1989 for analysis at the Intermountain Research Station laboratory in Moscow, ID. The samples were collected from existing plantation and orchard sites on a grid system. The analysis included soil moisture (field capacity and wilting point) and nutrients (N, P, K, S, Ca, Mg, Mn, Fe, and Cu).

Modal soils have been identified, described, and sampled for the Dry Creek site. These samples have also been analyzed for soil moisture and nutrients plus CEC, base saturation, pH, and particle size distribution. The nutrient analysis of existing sites will provide baseline data for the development of fertilizer regimes for each plantation and orchard.

SOIL MOISTURE MANAGEMENT

Conifers growing under natural conditions in the Northwest complete their height growth in the late spring and early summer when adequate soil moisture is available from seasonal precipitation and snow melt. Trees then set bud and height growth ceases during the summer drought, which usually is at the time of high evaporative demand, high air temperature, and low soil moisture. The plant moisture stress level reached during this period prevents second flushing, and the trees enter the dormancy cycle.

Irrigation in seed orchards and early selection trials is used to promote plant growth, protect the seedlings from frost damage, augment other cultural practices such as fertilization, and harden seedlings. Improper application of water in plantations could result in growth reduction, lowering of cold hardiness (Douglas-fir), lowering root and shoot growth, and the reduction of seedling vigor, all of which can increase mortality.

Regional direction (Forest Service Handbook 2409.26g) states that soil moisture in the root zone of early selection trials should be maintained at less than 5 bars for all species except lodgepole pine, which should be maintained at less than 8 bars. Soil moisture in seed orchards should be maintained at or above 50 percent of field capacity for all species.

After the areas have been mapped and soil moisture release curves produced, tensiometers and gypsum blocks are installed to monitor soil moisture. This makes it possible to use irrigation to keep soil moisture tensions within the desired ranges.

SOIL FERTILITY MANAGEMENT

Soil fertility will be managed through soil nutrient monitoring and fertilization. The objective will be to provide seedlings with a maintenance level of nutrients to maximize plant vigor while keeping vegetative growth at normal levels. The intent is to not promote vegetative growth. Optimal nutrient levels for Douglas-fir have been reported in various papers and presentations (Boyer 1982, 1989; Mandzak 1988).

Soil and foliar samples will be collected on all EST's and seed orchards to monitor nutrient status. Soil samples will be collected at 2- to 3-year intervals to ensure that soil nutrient levels are being maintained within established guidelines. The soil sampling will also be used to track the location of nutrients in the soil profile. The foliar analysis will be made each year to track nutrient status in the trees. The results of both sampling programs will be used to further refine fertilizer application rates to provide the nutrients that are limiting growth. In the future these sampling techniques will be used to develop fertilizer rates to promote pollen flowers and cones for seed production.

CONCLUSIONS

Soil management is a critical part of the management of tree improvement areas. Soils information is needed at several stages of site development. Soil mapping will be needed initially to be sure the site is suitable for a seed orchard or early selection trial. An assessment should also be made to determine the impacts on the soil, such as compaction, displacement, and mixing, from the clearing operations.

Site-specific mapping should be done prior to establishment of each seed orchard and early selection trial plantation. This detailed mapping can locate small inclusions significant to management.

Sampling, testing, and monitoring should be done by soil type. Soil moisture and temperature sensors should be established at sites that are representative of that soil type.

Soil moisture monitoring is necessary to ensure that sites meet the guidelines established for seed orchards and early selection trial plantations. Soil fertility monitoring and additions of fertilizer can ensure that nutrition is adequate to meet tree improvement area needs.

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PLANT SUCCESSION WITHIN MANAGED GRAND FIR FORESTS OF NORTHERN IDAHO

Pat Green and Mark Jensen

Land classification based on potential vegetation (for example, habitat types) has been applied in resource management in the Northern Rocky Mountains since the 1950's (Wellner 1989). Increasingly refined classes have provided managers with information on productivity, site limitations, and general vegetation trends after disturbance (Cooper and others 1987; Daubenmire 1952; Daubenmire and Daubenmire 1968). Recent studies have characterized successional pathways within habitat types to describe plant community development and change over time after disturbance (Arno and others 1985).

In February 1987, the Vegetation Coordinating Committee of the Northern Region, Forest Service, U.S. Department of Agriculture, recommended that a successional study be initiated in the grand fir/wild ginger habitat type (*Abies grandis/Asarum caudatum*). High costs of timber management and apparent problems associated with conifer reforestation in a vegetation complex called the grand fir mosaic necessitated this recommendation. An estimated 130,000 hectares (325,000 acres) of this complex occur in the Clearwater and Nez Perce National Forests of northern Idaho, which consists of closed and open stands of grand fir and Engelmann spruce (*Picea engelmannii*). Forest openings within the complex are commonly occupied by Sitka alder (*Alnus sinuata*), bracken fern (*Pteridium aquilinum*), and western coneflower (*Rudbeckia occidentalis*). These species may also be important understory components in timbered stands.

The grand fir mosaic occupies an ecological zone between climax western redcedar (*Thuja plicata*) and subalpine fir (*Abies lasiocarpa*) at elevations between 1,370 and 1,675 meters (4,500 and 5,500 feet). Conifer regeneration is generally sparse and localized, establishing where conifers occurred before site disturbance and at approximately the same number of trees per hectare. Seral tree species (for example, Douglas-fir [*Pseudotsuga menziesii*], western larch [*Larix occidentalis*]), common in other grand fir habitat types, are generally absent. Pocket gopher (*Thomomys* spp.) populations are high in many undisturbed stands and expand rapidly when the forest tree canopy is removed through timber harvest. Mosaic areas have high structural diversity, and commonly provide big-game hiding cover and summer habitat.

OBJECTIVES

The primary objective of this study was to describe successional plant community development within the grand fir/wild ginger habitat type and associated sites where wild ginger is a common understory species (wild ginger union). A secondary objective was to display range, wildlife, timber, and fire management resource values for each of the plant communities described.

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METHODS

The timber stands sampled in this study were identified with the help of local Forest Service personnel. Approximately 400 stands were sampled between 1986 and 1989 in the Nez Perce and Clearwater National Forests of northern Idaho. These stands represented different ages since treatment and different treatment types and intensities, and occurred across the geographic range of the wild ginger union. Plots were paired so that both untreated stands and adjacent treated stands were sampled.

Plots were located subjectively without preconceived bias following Mueller-Dombois and Ellenberg (1974). Where vegetation response appeared highly variable within treated stands, additional sample plots were established to represent seral plant communities and correlated to type and intensity of disturbance. For example, a clearcut, tractor-skidded, and broadcast-burned stand will normally display differing patterns of machine compaction, scarification, and burn intensity, which results in differences in vegetation response. The 0.04-hectare (1/10-acre) macroplots used in sampling were located to represent a uniform treatment type and associated vegetation expression (for example, one plot to represent each of the above disturbances).

Within each macroplot, foliar canopy cover for all herbaceous and woody plants was visually estimated. Canopy cover was recorded by 10 percent classes except for low-cover classes, which were denoted as T for less than 1 percent and P for 1 to 5 percent. Mean height of each species was estimated for animal hiding and thermal cover calculations. Understory species sampling methods utilized in this study follow those described in Chapter 4 of the Ecosystem Classification Handbook of the Forest Service, Northern Region (Hann and others 1988). Plant names and associated six-letter alphanumeric codes are shown in table 5.

A variable-radius plot (centered on the 0.04-ha macroplot) was established for measurement of all tree species 12.7 cm (5 inches) or more in diameter at breast height. Species, diameter, crown class, tree class, and crown ratio were recorded by tree size class. Additionally, age, height, and radial growth were recorded for selected growth sample trees. Smaller trees were tallied by species and size class in five 0.001-ha (1/300-acre) microplots. Tree origin (planted or natural) was noted and gopher mounds and cattle scat were tallied by microplot. The tree sampling methods utilized follow Forest Service, Northern Region stand exam procedures (USDA FS 1989).

Soil characteristics were described at each fixed plot using methods outlined in the Soil Survey Manual (USDA SCS 1981). Soils were described to the family level of Soil Taxonomy. Evidence of compaction, soil mixing, displacement, and the presence of charcoal was also noted.

Site features recorded at each macroplot included: elevation, landform, slope aspect, slope shape, slope gradient, slope position, and ground cover. Summary vegetation features included: tree basal area, canopy cover by lifeform, annual production by lifeform, and woody fuel loading. Disturbance type, intensity, and date were recorded for the following types of disturbance: animal use, wildfire, timber harvest system, logging slash and site preparation method, regeneration system, and grazing. This information was obtained from Forest Service stand-treatment records, which were modified where necesssary to correspond with observed plot conditions. Data analysis and resource value calculations were facilitated by ECOPAC software developed by the Ecosystem Management Group of the Forest Service, Northern Region (Keane and others 1988).

The level of ecological classification (Hann and others 1988) appropriate for describing plant community successional response was determined by contrasting seral community floristics by treatment type and ecological classification grouping (habitat type, habitat type phase, and site type). Given that vegetation response is a function of both site and disturbance variables, it was considered necessary to isolate community type variability due to site variables (for example, elevation, aspect, plot position, ash cap depth, soil texture) first in this analysis. Once reasonably uniform site potentials were elucidated from the data, plant successional response to disturbance type and intensity was determined.

Successional plant community pathways were developed by site potential type based on stand structure and species composition following Arno and others (1985, 1986). Five structural stages were recognized: (1) herbaceous/seedling, (2) shrub/sapling, (3) pole, (4) mature seral forest, and (5) old-growth forest. Plots within each site potential type and structural stage were aggregated into community types based on floristic similarity using TWINSPAN (Hill 1979). All plots in a group were examined for similarities in time since disturbance, treatment type or intensity, or additional site features that could improve vegetation response prediction.

Successional pathways were described for each site type by arraying sample plots with similar floristics and similar treatment types and intensities into developmental sequences. The resulting pathways were examined for consistency with known trends in canopy coverage of successional species. Posttreatment community types were contrasted with pretreatment stand composition, which assisted in clarifying successional relationships.

RESULTS

In most cases, analysis at the habitat type or habitat type phase level of ecological classification did not provide a site potential classification that could be used to predict plant community development after disturbance. Further stratification of the phase of the habitat type by various site factors (for example, aspect, elevation, substrate, and soil moisture) was required. This level of ecological classification (site type) proved most useful in predicting plant successional response following treatment. Site types of the grand fir/wild ginger habitat type identified in this study are presented in table 1. An example of successional pathways within a site type is presented in figure 1. Complete descriptions of site types and seral plant communities within the study area are available from the authors.

Each of the plant communities identified for a site type possesses unique values for a variety of resource uses (table 2). Forage production, pocket gopher activity, fuel loadings, and big-game hiding cover are closely associated with the plant species composition of a site, which in turn changes with time since disturbance. Table 2 displays

how resource values change following clearcutting and low-intensity broadcast burning.

The ability to predict vegetation response to treatment allows the land manager to more fully determine the long-term effects of activities on the forest landscape. Once a timber stand has been assigned to an appropriate site type, vegetation response can be evaluated for alternative site treatments. Plant community composition and associated resource values may then be compared to choose a treatment that enhances values that best meet management objectives. These points are illustrated in table 3, which presents properties associated with two early seral plant communities of the grand fir/wild ginger/wild ginger-cold, dry site type. High-intensity mechanical scarification on this site type results in a strawberry (*Frageria vesca*)-thistle (*Cirsium* spp.) dominated plant community, which takes over 10 years to develop desired tree stocking levels. The high gopher activity associated with this community type requires that large expenditures of money be invested to obtain desired stocking levels (repeated gopher treatment and tree planting are required). Low-intensity mechanical scarification on this ecological type results in the development of a beargrass (*Xerophyllum tenax*)/thistle plant community, which requires less investment for tree establishment and, therefore, best meets management objectives.

The ability to predict vegetation response following treatment is a function of the level of ecological classification used in describing the treated area. In this study, different plant communities were associated with the same type of disturbance (table 4) when sites were described at the habitat type or phase of habitat type levels of ecological classification (Cooper and others 1987). Consequently, accurate predictions of vegetation response were not possible at higher levels of ecological classification. Defining the site type of a stand, however, facilitated reasonable prediction of vegetation response by type and intensity of silvicultural disturbance.

Table 1—Site types of the grand fir/wild ginger habitat type (species names associated with the abbreviations shown are presented in table 5)

Habitat type phase	Site type
ABIGRA/ASACAU/ASACAU	1. Low elevation, dry 2. LINBOR, sandy substrate 3. XERTEN, cold, dry 4. ACTRUB or ADEBIC, moist
ABIGRA/ASACAU/MENFER	1. LINBOR, dry 2. XERTEN, cold 3. Moist
ABIGRA/ASACAU/TAXBRE	No site types defined

CONCLUSIONS

The habitat type classification of Daubenmire (1952) has been modified in recent years to describe 49 habitat types and 60 habitat type phases in northern Idaho (Cooper and others 1987). This level of ecological classification provides

Conditions or treatments	Structural stage: Herbaceous	Sapling	Pole	Mature seral	Old growth
WF-LI					ABIGRA-ACEGLA / ASACAU-COPOCC
WF-MI	? - - - - - - - -	- - - - - - - -	- - - - - - - -	> ABIGRA-PSEMEN / ASACAU-COPOCC	ABIGRA-ACEGLA / ASACAU-COPOCC
CC, DP-LI, MI	EPIANG-MERPAN - - - -	- - - - - - - -	>?		
CC, DP-MI	MERPAN-RUDOCC - - - -	- - - - >?			
CC, SD, DP-HI	RUDOCC-BROVUL / FRAVES - - - -	- - - - - - - -	> ABIGRA-PSEMEN / ASACAU-ACTRUB - - - -	- - - - - - - -	>ABIGRA-ACEGLA / ASACAU-COPOCC
CC, BB-LI	RUMACE/PHAHET - - - -	> RUBPAR-SYMALB / ASACAU-PHAHET - - - -	- - - - - - - -	>ABIGRA-PSEMEN / ASACAU-COPOCC	ABIGRA-ACEGLA / ASACAU-COPOCC
CC, BB-MI SD, JP-MI	PTEAQU - - - -	> ACEGLA-RUBPAR - - - -	- - - - - - - -	>ABIGRA-PSEMEN / ASACAU-COPOCC	ABIGRA-ACEGLA / ASACAU-COPOCC
CC, BB-HI	EPIANG / FRAVES-MITSTA - - - -	> CEAVEL-RIBVIS / MITSTA - - - -	- - - - - - - -	>ABIGRA-PICENG / RIBVIS-ASACAU	ABIGRA-ACEGLA / ASACAU-COPOCC

DISTURBANCE CODES:
WF, Wildfire; CC, Clearcut; SW, Shelterwood harvest; BB, Broadcast burn; JP, Jackpot burn; DP, Dozer-pile slash; LI, Low intensity; MI, Moderate intensity; HI, High intensity.

Figure 1—Successional pathways within the ABIGRA/ASACAU/ASACAU-Moist site type. Species abbreviations are described in table 5.

Table 2—Resource value ratings associated with seral plant community development following clearcutting and low-intensity broadcast burning on the ABIGRA/ASACAU/ASACAU-Moist site type

Resource value rating	Community type		
	ACEGLA-RUBPAR	ABIGRA-PSEMEN ASACAU-COPOCC	ABIGRA-ACEGLA ASACAU-COPOCC
Structural stage sequence	Shrub/sapling	Pole	Mature seral
Age (years)	15-20	100-120	120-150
Basal area (ft²/acre)	2	230	257
Tree cover (percent)	2	58	63
Shrub cover (percent)	30	20	17
Forb cover (percent)	57	25	33
Graminoid cover (percent)	8	2	3
Big-game hiding cover (percent)	29	48	28
Cattle forage (kg/ha)	805	615	682
Elk forage (kg/ha)	1,042	902	1,053
Gopher forage (kg/ha)	856	486	687
Gopher (mds/ha)	1,151	37	457
Fuels (t/ha of >1-inch material)	108	19	72
Fire spread (m/min during dry conditions)	8.4	1.2	1.2
Flame scorch height (m)	2.3	0.5	0.5
Exposed mineral soil (percent)	10	4	5

Table 3—Comparison of two early seral plant communities associated with different types of disturbance on the ABIGRA/ASACAU/ASACAU-XETE-Cold, dry site type

Treatment	Plant community type	
	XERTEN-CIRSIU	FRAVES-CIRSIU
	Clearcut or seed tree; low to moderate intensity scarification	Clearcut or seed tree; high-intensity scarification
Age	2-10	2-25
Trees/ha	1,693	2,965
Dominant tree age (years)	4	5
Exposed mineral soil (percent)	17	24
Cattle forage (kg/ha)	486	423
Gopher forage (kg/ha)	775	785
Gopher (mds/ha)	457	1,150
Species richness	28-51	48-51
Years to attain desired tree stocking	<10	>10

general information on productivity and climatic characteristics; however, it was not specific enough to predict vegetation response in this study.

The site type is a useful refinement of the habitat type when prediction of plant successional response is an objective of classification. Within a site type, plant community development after disturbance tends to proceed at rates and directions that can be predicted. Predictions of species composition and associated resource values assist land managers in evaluating whether proposed activities will result in desired interim and future conditions of the forest landscape.

The ability to predict plant community response is critical to understanding the effects of management practices on western wildlands. The site types and plant successional pathways described in this study of the grand fir/wild ginger habitat type of northern Idaho provide land managers an efficient method of assessing the cumulative effects associated with multiple-use management.

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Table 4—Characteristics of two early seral plant communities following similar disturbance on two site types of the ABIGRA/ASACAU/ASACAU habitat type phase. The plant community descriptions presented represent 20 years of vegetation development following clearcutting and high-intensity broadcast burning

Item	Site type	
	Moist	Cold, dry
Dominant trees	PSEMEN	ABIGRA
Dominant shrubs	ACEGLA SYMALB	VACGLO
Dominant forbs	AREMAC CIRALP MERPAN SMISTE THAOCC	CIRARV CIRVUL XERTEN
Ferns	PTEAQU	—
Tree cover (percent)	2	4
Shrub cover (percent)	30	3
Grass cover (percent)	8	15
Forb cover (percent)	57	31
Fern cover (percent)	13	0
Trees/ha	635	1,693
Tree basal area (m²/ha)	0.5	2.8
Gopher (mds/ha)	1,151	3,010
Elk hiding cover (percent)	29	9
Cattle forage (kg/ha)	905	487
Elk forage (kg/ha)	1,043	871
Gopher forage (kg/ha)	696	776

Table 5—Plant names and associated alpha codes used in this paper

Plant name and authority	Forest Service, Northern Region, alpha code
Abies grandis (Dougl. ex D. Don)	ABIGRA
Abies lasiocarpa (Hook.) Nutt.	ABILAS
Acer glabrum Torr.	ACEGLA
Actaea rubra (Ait.) Willd.	ACTRUB
Adenocaulon bicolor Hook.	ADEBIC
Alnus sinuata (Regel) Rydb.	ALNSIN
Arenaria macrophylla Hook.	AREMAC
Asarum caudatum Lindl.	ASACAU
Bromus vulgaris (Hook.) Shear	BROVUL
Ceanothus velutinus Dougl. ex Hook.	CEAVEL
Circaea alpina L.	CIRALP
Cirsium spp.	CIRSIU
Cirsium arvense (L.) Scop.	CIRARV
Cirsium vulgare (Savi) Tenore	CIRVUL
Coptis occidentalis (Nutt.) Torr. & Gray	COPOCC
Epilobium angustifolium L.	EPIANG
Fragaria vesca L.	FRAVES
Mertensia paniculata (Ait.) G. Don	MERPAN
Mitella stauropetala Piper	MITSTA
Phacelia heterophylla Pursh.	PHAHET
Picea engelmannii Parry ex Engelm.	PICENG
Pseudotsuga menziesii (Mirbel) Franco	PSEMEN
Pteridium aquilinum (L.) Kuhn	PTEAQU
Ribes viscossimum Pursh.	RIBVIS
Rubus parviflorus Nutt.	RUBPAR
Rudbeckia occidentalis Nutt.	RUDOCC
Smilacina stellata (L.) Desf.	SMISTE
Symphoricarpos albus (L.) Blake	SYMALB
Taxus brevifolia Nutt.	TAXBRE
Thalictrum occidentale Gray	THAOCC
Thuja plicata Don ex D. Don	THUPLI
Vaccinium globulare Rydb.	VACGLO
Xerophyllum tenax (Pursh.) Nutt.	XERTEN

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AN INTEGRATED APPROACH FOR RIPARIAN INVENTORY

Mark Jensen and Pat Green

Riparian and aquatic ecosystems comprise a relatively small fraction of National Forest lands; however, their value for a variety of resources uses is considerable. Increased interest in management of these ecosystems dictates that an integrated systematic approach to riparian description be employed if the following questions are to be addressed:

1. What types of riparian areas do we manage?
2. What is the location and extent of such areas?
3. How do different types of riparian areas respond to management?
4. What is the existing condition of our riparian areas?
5. What is a reasonable "desired future condition" for such areas?
6. What types of treatment are appropriate if desired future conditions for riparian areas are to be achieved?

Current information for these ecosystems has been generated from various inventories of existing condition (for example, timber survey, fishery survey, best management practice monitoring, water quality monitoring). A description of riparian and aquatic ecosystem site potential is required if existing information is to be utilized in answering the questions posed earlier. To meet management needs, this site potential description must be hierarchical in design and integrate both riparian and aquatic ecosystem components with upland systems. The methods of riparian map unit design and sampling (Jensen 1990) utilized by the Northern Region, Forest Service, accomplish these needs. This paper presents an overview of those methods.

SITE POTENTIAL MAP UNIT DESIGN

A description of site potential is required if assessments of riparian condition are to be made; consequently, site potential map units are commonly the first item developed in riparian inventory. Riparian site potential map units may be developed at different scales of resolution dependent upon management needs.

Three levels of map unit design are presented, which are approximately equivalent to those utilized by other Forest Service Regions in riparian inventory (USDA FS 1990, 1989). These levels range from rapid, office-based procedures (level 1) to detailed, field-verified procedures (level 3). Mapping scale, classification, and map use are all influenced by the level of map unit design utilized in riparian inventory (table 1). Following is a brief description of the three levels of riparian site potential map unit design utilized in the Northern Region.

Level 1 map unit descriptions are developed in the office utilizing existing sources of information (for example, topographic maps, aerial photos, land systems inventory). Basic climate, geology, landform, and soil parameters that affect stream sediment delivery and transport, upland slope hydrology, and upland potential vegetation are synthesized at this level of mapping. Map unit descriptions are stored in various databases for characterization of watersheds and stream reaches. Level 1 map units are commonly delineated on 1:100,000 to 1:24,000 topographic base maps for future geographic information system application.

Level 2 map unit descriptions and databases are developed from field reconnaissance-level surveys of stream reaches. Stream channel morphology, bank stability, sedimentation, and potential riparian vegetation-soil relationships are examples of information contained within such riparian descriptions. Level 2 map units are commonly delineated as line segments or polygons on 1:24,000 or larger base maps and are used to characterize stream reach segments.

Level 3 map unit descriptions are developed based upon project level objectives. The following surveys are examples of data sources from which level 3 map unit descriptions may be developed: order 1 soil survey, fishery habitat survey, riparian site type survey. Level 3 map units are delineated on 1:24,000 (or larger) base maps and are commonly used to describe, in detail, specific components of level 2 map units (for example, soil taxa and stream habitat).

HIERARCHICAL MAP UNIT COMPONENTS

Nine hierarchical components (table 2) are utilized in describing riparian and aquatic map units following Jensen and others (1989). These hierarchical components describe the landscape by progressively more refined criteria, from which increasingly specific interpretations of the landscape may be derived. The various scales of description facilitated by these components allow regional to site-specific assessments of site potential. Site-specific information used to describe lower component levels can also be aggregated to develop interpretations for higher component levels.

CONCLUSIONS

Site potential map unit descriptions provide a conceptual categorization of the landscape useful in assessing riparian condition (the value of a riparian area for a particular set of resource uses relative to its potential). Given the fact that many of the components of levels 1 and 2 map units do not change in response to management activities (for example, landform, geology, and valley bottom gradient), these map units may be used to consistently describe similar environments regardless of disturbance history.

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Table 1—Comparison of riparian/aquatic site potential map unit design levels

Item	Map unit level 1	2	3
Level of application	Regional assessments, forest planning	Forest and project planning	Project planning
Minimum mapping scale	1:100,000-1:24,000	1:24,000	1:24,000 or more detail as needed
Components	Ecoregion/area	Valley bottom subtype	Appropriate to need
	Geologic group	Rosgen stream type	
	Landtype association	Fishery habitat	
	Valley bottom Landtype	Riparian vegetation and soil composition	
	Upland vegetation and soil composition	Stream channel and bank characteristics	
Common application	General watershed description; synthesis of existing information;	General stream reach description	Specific stream reach description
	describe sedimentation	Describe stream system equilibrium	Baseline monitoring
	Describe high-water yield areas	Describe stream sedimentation	Quantify stream condition
	Describe unstable watersheds	Describe high-value fisheries	Quantify riparian condition
	Describe high-energy watersheds	Describe potential riparian vegetation	Validate forest planning standards and guidelines

Once the site potential map unit has been determined for a given reach, its current features are contrasted to other managed expressions of similar site potential to determine its condition. Assessments of riparian condition are meaningless without an understanding of site potential; consequently, site potential map unit development should be the first process undertaken in riparian inventory. An example of a level 2 site potential map unit description (developed by Green) is provided in appendix A.

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Table 2—Hierarchical components of riparian map unit design. Ecoregions through valley bottom landtype are used in designing Level 1 map unit descriptions and are not significantly altered by management practices. Valley bottom subtypes through landform are used in designing Level 2 map unit descriptions and are not commonly significantly altered by management practices. Vegetation type through stream state are used in designing Level 3 map unit descriptions and may be significantly altered by management practices

Component	Description
Ecoregion	An area determined by similar physiographic province and dominant lifeform (Bailey 1978).
Ecoregion area	A portion of an ecoregion with similar climate and natural vegetation. Factors such as seasonal storm pattern, annual temperature, and biogeography are utilized in describing ecoregion areas.
Geologic group	A portion of an ecoregion area with relatively homogeneous parent materials, distinguished from surrounding groups by structure, degree of weathering, dominant size-fractions of weathering products and water-handling characteristics (for example, porosity, permeability, runoff potential). Geologic groups include both uplands and bottomlands, and are commonly derived from Forest Service land system inventory subsection maps (Wertz and Arnold 1972).
Landtype association	A portion (or all) of a geologic group that is distinguished by a dominant geomorphic process (for example, glacial, fluvial, alluvial, lacustrine). Landtype associations include both uplands and bottomlands and are commonly described in Forest Service land system inventories.
Valley bottom landtype	A portion of a landtype association distinguished by landform and position. Landtypes correlate with associations of soils and plant communities, and constitute the most refined level of land systems inventory conducted in the Northern Region of the Forest Service.
Valley bottom subtype	A portion of the valley bottom landtype distinguished by fluvial geomorphic properties that influence the manner in which water and sediment move through the system. Valley bottom subtypes are commonly distinguished by significant changes in Rosgen (1985) stream type.
Landform	A portion of the valley bottom subtype with distinctive morphology (for example, channel, flood plain, levee, and alluvial fan). Landforms are usually highly correlated with soils and vegetation community distribution.
Vegetation type	A distinctive plant community that is usually identified by lifeform class (for example, forested, shrub, herbaceous) and dominant plant species in the overstory and understory canopies.
Stream state	The existing state of the stream relative to its equilibrium condition within the valley bottom subtype. Factors such as stream downcutting or widening are considered in assessing stream state.

APPENDIX A: EXAMPLE OF A LEVEL 2 RIPARIAN MAP UNIT DESCRIPTION DEVELOPED BY PAT GREEN, NEZ PERCE NATIONAL FOREST

C6 CHANNELS, GRANITIC SOURCE AREA
CARROS-CARAQU/DANINT-ABLA/CACA VEGETATION COMPLEX

Beaked sedge-Aquatic sedge/timber danthonia-Subalpine fir/bluejoint reedgrass

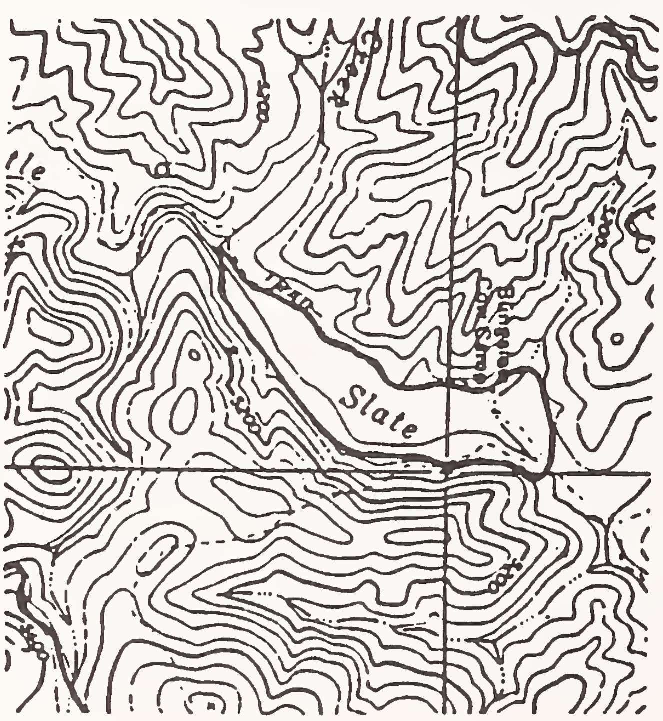
ENVIRONMENTAL DESCRIPTION

Elevation (ft):	Range = 5000 - 6000 Mean = 5170
Landforms:	Narrow to moderately broad (100-600 feet wide) nearly flat low relief alluvial basins, including streamsides and depressions, low floodplains, old flooplains and rolling uplands with lower slopes within the stream influence zone.
Stream gradient:	Range = .5 - .9 Mean = .7
Microclimate:	Frost pocket
Corresponding National Wetlands Inventory Map Unit:	PEMIC
Corresponding General Forest Riparian Type:	F4

LOCATION AND RIPARIAN LANDFORMS

This riparian type commonly occurs on nearly flat, low lying alluvial floodplains at middle to upper elevations in the Florence basin and other geologic zones dominated by quartz monzonite of the Idaho batholith. Surrounding uplands are low rolling hills and moderately steep rolling uplands.

The map below shows the typical position of this riparian reach in the landscape.



CHANNEL DESCRIPTION

In undisturbed systems this C6 channel (Rosgen 1985) is narrow, deep and meandering, with a low gradient (<.9 %), high sinuosity (>2.5) and width/depth ratio of 3 or less. Channel materials are gravels and sands, and the stream is deeply entrenched and poorly confined, with seasonal flooding.

SOIL DESCRIPTION

Sandy Typic Cryofluvents and Cryopsamments occur in low floodplains in this riparian type. Organic surface layers up to 20 inches thick may be present in the wettest low lying positions. Soils with these layers are Cryohemists.

Medial over sandy Andeptic Cryofluvents and Typic Cryopsamments are on higher floodplains. Sandy Andic Cryochrepts are on upper terraces and lower slopes of adjacent uplands.

Range In Soil Characteristics

Sandy Typic Cryofluvents and Cryopsamments have dark brown sandy loam surface layers 5 to 7 inches thick that overly dark gray to light brownish gray loamy sands and sands.

Depth to gleying is 5 to 24 inches. The seasonaly low water table is 0 to 40 inches, and the water table is near or slightly above the soil surface much of the growing season.

Andeptic Cryofluvents have loam or silt loam surface layers and grayish brown subsoils and substrata. The seasonaly low water table is 30 inches or more.

Andic Cryochrepts have dark brown or strong brown silt loam or sandy loam loess influenced surface layers and brown sand or loamy sand substrata. The seasonal low water table is 40 inches or more. In lower slope positions, subsoil and substrata horizons may have few to common strong brown mottles.

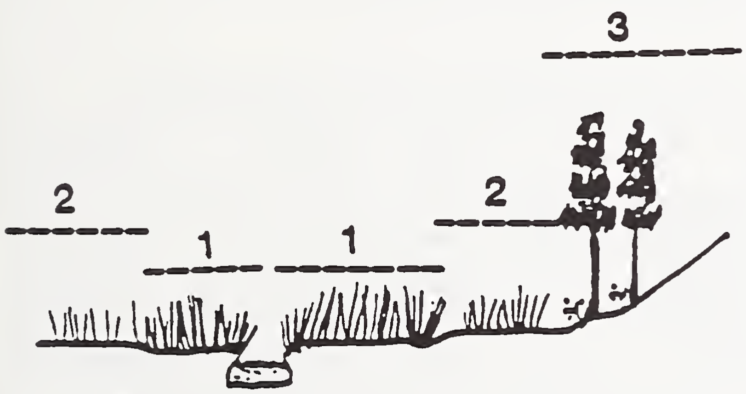
VEGETATION DESCRIPTION

The natural vegetation of this riparian type is a complex dominated by sedge and grass plant communities. These plant communities are illustrated in the diagram below.

1) Carex rostrata is the dominant plant species along low stream margins and in depressions. This plant community occupies 45 percent of the riparian reach.

2) The Carex aquatilis/Danthonia intermedia plant community is on drier sites. This plant community occupies 35 percent of the reach.

3) The Abies lasiocarpa/calamagrostis canadensis habitat type occurs on low terraces and lower slopes of adjacent uplands. Pinus contorta and Vaccinium scoparium usually dominate these stands. This habitat type occupies about 20 percent of the reach (i.e., riparian area and riparian influence zone).



MANAGEMENT

Herbage production in the C. rostrata plant community is high (1700 to more than 3500 pounds per acre) but utilization of C. rostrata is low. C.aquatilis is only lightly utilized. Calamagrostis canadensis, Agrostis tenuis, Phleum alpinum and Triseum wolfii are heavily utilized when the site is dry enough to be accessible to cattle. As much as 80 percent of the available forage from these species is used on the drier sites that are accessible for much of the growing season.

Herbage production in the C. aquatilis/D. intermedia community is moderate (1200 to 1800 pounds per acre). C. canadensis and P. alpinum are heavily utilized. This community is on drier sites than the C. rostrata community, is available for cattle grazing for longer periods, and can suffer greater trampling damage. It is less often adjacent to streambanks, and grazing impacts are less likely to affect stream channels.

The Abies lasiocarpa/Calamagrostis canadensis habitat types are on drier sistes that offer shade and resting sites as well as areas of forage. Production is moderate (1000 to 1600 pounds). These areas are heavily used by cattle, deer, and elk. Even X. tenax and V. scoparium are browsed and hedged. Palatable grasses, including P.alpinum, C. aquatilis, C. canadensis, and Agrostis exerata are heavily grazed. Trifolium repens and Fragaria vesca tend to increase their cover on compacted sites.

Along the channel, C. rostrata and C. aquatilis form a dense root mat that is highly resistant to erosion. Streambanks readily undercut below this root mass, especially in this geologic material where thin, finer textured surface soil layers overlay sands. This process forms pools with good shade and hiding cover for resident fish. However, the banks will slough with trampling. Once banks are damaged and stream dynamics are changed, renewed stream downcutting may lower the water table and make maintainance of the sedge plant community difficult.

Within elevations of 4000 to 6500 feet, and where water tables are close to the surface, C. rostrata is a good bank restoration species (Hansen and others 1988).

Pool quantity and quality can easily be damaged in this riparian type due to the high proportion of sands in channel sediments and noncohesive bank materials which are susceptible to trampling and sloughing. Trees and shrubs are not readily available for instream cover or debris reruitment. Timber management on adjacent forest sites can influence this type through raising the water table temporarily as a consequence of timber harvest, and introducing additional sand size sediment through road construction and harvest operations.

Grazing and wildlife management influence the condition of this riparian type. Management should maintain natural channel and vegetation characteristics and low stream width/depth ratios which help sustain flushing flows to move excess sands out of the system.

This channel type is very sensitive to vegetation change. Impacted areas commonly change to C3 or C4 channels, which display unstable banks, stream downcutting and a drop in the local water table. Under uncontrolled season long grazing systems, soils are compacted and native plants are damaged. This process results in bank damage and soil erosion.

Since C. rostrata is not highly palatable, more palatable associated species such as Calamagrostis canadensis should be used to assess utilization levels. Proper levels of grazing range from light to moderate (40% utilization).

Salix boothii at lower elevations and Salix wolfii at higher elevations may be planted to assist in bank stabilization by providing deeper rooting material. Physical bank protection measures, such as fences or log barriers, may be necessary to prevent livestock trampling of banks.

Under badly degraded conditions, appropriate fish habitat improvement structures include bank placed boulders, floating log covers or submerged log shelters located on straight reaches, and reestablishment of deep rooted vegetation along streambanks.

GENERAL PLANT COMMUNITY CHARACTERISTICS

The following is a series of tables which summarize basic ecosystem data collected at 6 sites used to describe this riparian reach.

Average Percent Canopy Cover by Life Form for Each Plant Community

Life Form	CARROS	CARAQU/ DANINT	ABILAS/ CALCAN
Tree	1	0	80
Shrub	0	0	30
Graminoid	87	80	30
Forb	20	20	30
Fern	0	0	.5

The following table displays constancy and average canopy cover values for important plant species by plant community.

Species	CARROS	CARAQU/ DANINT	ABILAS/ CALCAN
Trees			
ABILAS			100(2.2)
PINCON			100(38.6)
Shrubs			
VACCES			80(3.2)
VACSCO			100(18.4)
Graminoids			
AGREXA			70 (3.5)
AGRSCA		100(3.4)	
AGRTEN	100(4.5)		
CALCAN	100(7.7)	100(3.0)	100(3.2)
CARAQU	100(17.7)	100(7.2)	100(2.4)
CARINT	100(1.3)		
CARROS	100(31.0)		
DANINT		100(13.2)	
JUNENS	100(3.7)		
MUHFIL		100(6.4)	
PHLALP	100(1.3)		100(1.4)
Forbs			
ANEPIP			100(1.2)
ANTMIC			100(1.4)
ASTOCC	100(2.2)	100(5.0)	100(1.6)
COPOCC			100(1.2)
EPIWAT	100(1.3)		
FRAVES		100(6.3)	
SANSIT	100(4.5)		
TRIREP			80(5.6)
VIOLA	100(2.2)		
XERTEN			100(2.1)
Ferns/ Mosses			
MOSS	100(30.0)		

Production (lbs/ac/yr-dry wt.) by Plant Community

Life Form	CARROS	CARAQU/ DANINT	ABILAS/ CALCAN
Shrub	0-50	0	50-200
Graminoid	1700-3600	1200-1800	800-1200
Forb	60-200	100-300	50-200

Ground Cover Composition (%) by Plant Community

Component	CARROS	CARAQU/ DANINT	ABILAS/ CALCAN
Bare Soil/Gravel	0-10	0-3	0 - 10
Rock	0	0	0
Basal Vegetation	.5-10	1-10	1-10
Woody Debris	.5-10	0-1	10-30
Litter	20-70	60-80	50-80

RESOURCE VALUE RATINGS

The following is a listing of resource value ratings associated with the community types described in this riparian reach. Analysis employed follows methods outlined in Chapter 5 (ECODATA Data Bases and ECOPAC Analysis Software) of the Ecosystem Classification Handbook, U.S. Forest Service, Northern Region.

Wildlife Cover Table by Plant Community

Index	CARROS	CARAQU/ DANINT	ABILAS/ CALCAN
Hiding Cover (%) at 4.0 ft. View Height	0	0	27
Plant Cover, 40 ft. Index	0	0	63
Summer Thermal Cover Index	33	33	73
Winter Thermal Cover Index	0	0	42
Wind Blockage Index	0	0	4

Note: Index values range from 0 (low) to 100 (high).

Average Summer Forage Values (Lbs/ac/yr - dry wt.) by Plant Community

Animal Species	CARROS	CARAQU/ DANINT	ABILAS/ CALCAN
Cattle	1171	749	686
White Tail Deer	792	580	461
Elk	1082	815	689
Moose	885	616	557

Percent Elk-Cattle Forage Similarity by Season of Use and Plant Community Type

Season	CARROS	CARAQU/ DANINT	ABILAS/ CALCAN
Summer	96	99	94
Winter	92	80	92

Indices of Plant Community Diversity

Diversity Index	CARROS	CARAQU/ DANINT	ABILAS/ CALCAN
Structural Diversity	.19	.15	.82
Shannon-Weaver Index	1.09	1.20	1.08
Species Richness		37	3043

STREAM CHANNEL PROPERTIES

The following is a listing of stream channel properties associated with this riparian type. Analysis employed follows methods outlined in Chapter 5 (ECODATA Data Bases and ECOPAC Analysis Software) of the Ecosystem Classification Handbook, U.S. Forest Service, Northern Region.

Stream Channel Characteristics under Natural and Grazed Conditions

Property	Natural Conditions	Under Prolonged Season Long Grazing
Stream Order	2 - 4	Unchanged
Valley Bottom Gradient (%)	<3	Unchanged
Rosgen Classification	C6	C6 to C3 or C4
Bankful Width (ft)	5 - 15	5 - >15
Depositional Features	None	Point Bars
Sinuosity	1.5 - 2.0	1.3-1.8
Width/Depth Ratio	2.5	>3.0
Pool	70	10 - 20
Embeddedness %	50	30 - 60
Dominant Particle Size	S	FS
Glide	27	75 - 80
Embeddedness %	40	20 - 50
Dominant Particle Size	S	FS

Stream Channel Characteristics under Natural and Grazed Conditions (continued)

Property	Natural Conditions	Under Prolonged Season Long Grazing
Riffle	3	4 - 10
Embeddedness %	30	10 - 30
Dominant Particle Size	S	SG
Percent Undercut	80	40 - 80
Width Undercut (ft.)	1.1	.5
Stream Overhanging Plant Cover		
Trees	.5	0
Shrubs	.5	.5
Herbs	.5	.5 - 10
Streambank Plant Cover		
Trees	.5	0
Shrubs	.5	.5
Herbs	100	70 - 80
Streambank Ground Cover		
Bare Soil	1 - 3	10
Gravel	0	.5
Rock	0	.5
Litter	60	50
Wood	1 - 3	1 - 3
Moss	30	30
Basal Vegetation	10	10
Percent Stable Banks	80 - 90	60 - 80

A METHOD FOR PHOTOSYNTHETIC CALIBRATION OF PROCESS MODELS FOR CONIFER ECOSYSTEMS

A. Koehn and G. I. McDonald

This poster synopsis describes procedures to estimate calibration parameters for the assimilation portion of a process-based forest ecology model known as Forest-BGC (Running and Coughlan 1988). Calibration will facilitate site-specific simulation of key system processes involving carbon, water, and nitrogen cycles. We describe a procedure using a Clark-type oxygen electrode system to make laboratory estimates of maximum mesophyll conductance, light compensation point, and the Michaelis-Menton constant for light. The Michaelis-Menton constant we refer to in this paper is not exactly the same as the Michaelis-Menton constant of enzyme kinetics; rather it is the calculated value from nonlinear regression analysis.

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MATERIALS AND METHODS

Data collected from two trees in February 1990 were used to illustrate calibration procedures. Tree 123 was a control tree, shaded only by the greenhouse structure (35 percent full sun), and tree 144 was from the shade treatment which received 20 percent of the light of the control treatment.

Photosynthetic oxygen evolution rates were measured using the leaf-disc electrode system designed by Delieu and Walker (1981, 1983) and manufactured by Hansatech Instruments Ltd., England. The system consisted of a Clark-type electrode that measures oxygen polargraphically. Light was supplied by the Hansatech LS-2 unit consisting of a 100-watt photo lamp, a heat-reflecting mirror to reduce the infrared content of the emitted light, a system of lenses to produce a beam of light with low variation of intensity across the beam and little divergence from parallel, and three slots that accept 50-mm^2 neutral density filters to give precise control of the desired light intensity. The lamp housing was cooled by a high-output 12-v d.c. fan, which blows filtered air across the bulb and heat-reflecting mirror. Photon flux density of each light level was measured using an Omnidata Model ES-240 quantum sensor.

Oxygen evolution was measured at 20 ± 1 °C and saturating CO_2 (5 percent CO_2). The CO_2 concentration was controlled by a sodium carbonate/bicarbonate/borate buffer solution, which wets a capillary matting in the bottom of the chamber. Temperature control was provided by a controlled-temperature water circulator. Oxygen evolution was measured at eight different photon flux densities (PFD's): 30, 50, 95, 170, 230, 430, 820, and 1,500 µmoles/m^2/s (full sun for the summer averages approximately 1,600 µmoles/m^2/s). Dark respiration was measured with the lights off at the beginning of the sampling session. Five voltage readings were recorded with a Polycorder 700 data recorder in 300-microsecond intervals every 15 seconds for 3 minutes at each PFD and for dark respiration (fig. 1). The five voltage readings from each 15-second interval were averaged to provide one reading. Twelve readings were obtained over a 3-minute period at one PFD.

The electrolyte used for the electrode was one part saturated KCL solution, one part 0.4 M borate buffer at pH 9.0, and two parts 1.0 M sodium bicarbonate adjusted to pH 9.0 with 1 M sodium carbonate solution. The electrode was covered by the electrolyte, a 2 cm^2 piece of cigarette paper, and a teflon membrane of the same size.

The system was calibrated by injecting 1 cm^3 of air into the chamber and recording the change in voltage (the capillary matting already in the chamber). The system calibration calculations were:

1. correction for standard temperature and pressure:

   X = µmoles O_2/1 cm^3 of air at 20 °C.
   = 9.37 (P/Po) (273/T); P/Po = exp (–A/8,500),
   (A = altitude in meters)
   = 9.37 (0.91) (273/293)
   = 7.94 µmoles O_2/1 cm^3 of air at 20 °C.

2. µmoles O_2/mv = 7.94 µmoles O_2/mv during calibration.

Oxygen evolution rates were calculated using linear regression on the data points for the 3-minute readings for each photon flux density (fig. 2). Electrode sensitivity to temperature changes that occur with increasing light intensity was compensated for by completing a blank (tissue absent) run over all PFD's. Difference between the slopes of the regression lines (fig. 2) was the rate of O_2 evolved (or in the case of dark respiration, O_2 uptake). Needle area was measured using a high-resolution monochrome TV camera, a PCVISION*plus* Frame-grabber, and Java video analysis software.

Micromoles of O_2/m^2/s for the needle sample was calculated as follows:

1. µmoles/mv (value from calibration) * mv output/leaf area
   = µmoles/leaf area
2. corrected slope from regression of µmoles/leaf area vs. time
   = µmoles O_2/leaf area/second
3. (µmoles of O_2/leaf area (mm^2)/sec) * 1,000,000 mm^2/m^2
   = µmoles of O_2 evolved/m^2/s.

At low photon flux densities (up to 95 µmoles/m^2/s), the rate of photosynthesis is directly proportional to PFD and quantum yield (photon yield) is constant and maximal (Bjorkman and Demmig 1987). Quantum yield is a measure of the efficiency with which light is converted to stable photosynthetic products. On the assumption that plants use the same photosynthetic pathways and are equally efficient in converting photons into chemically bound energy, one would expect them to have identical quantum yields as long as the functional integrity remains intact (Bjorkman and Demmig 1987). We mention quantum yield because it is a potential quantitative indicator of the effect of stress on the photosynthetic system. Quantum yield is equal to the slope of the photosynthetic rate vs. PFD at low light levels. The light compensation point and dark respiration are also derived from the same linear regression analysis (figs. 3 and 4).

The method used to extract values of calibration parameters P_{max} and K_m from O_2 evolution data was a nonlinear regression analysis. Enzfitter (Leatherbarrow 1987) is a nonlinear regression program designed to analyze enzyme kinetics. The model for the anaylsis was the Michaelis-Menton enzyme kinetics equation and fit of actual to model was adequate (fig. 5). Photosynthesis values used in this analysis are net photosynthesis + dark respiration.

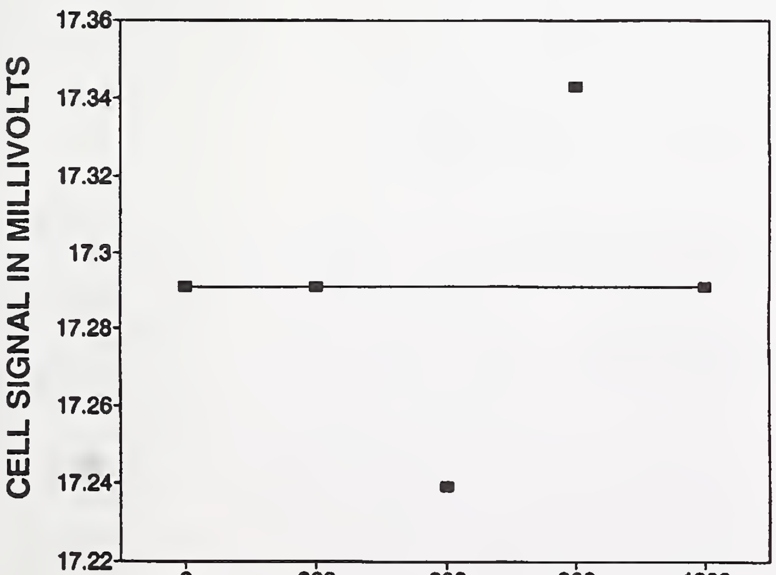

Figure 1—Five voltage readings shown in this graph illustrate the variation in the Polycorder 700 signal.

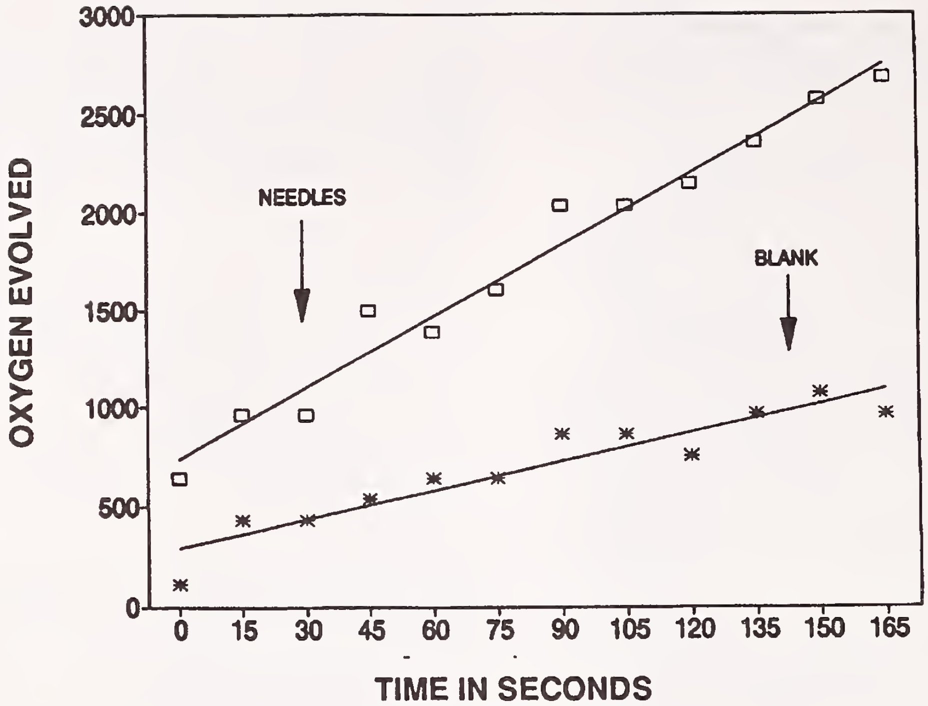

Figure 2—O_2 evolution rate for a 3-minute period at one PFD is calculated from the difference in slope (µmoles $O_2/m^2/s$ vs. time) between needle and blank. Each data point is composed of the mean of five readings (fig. 1). These calculations are made for all nine PFD's (including dark respiration).

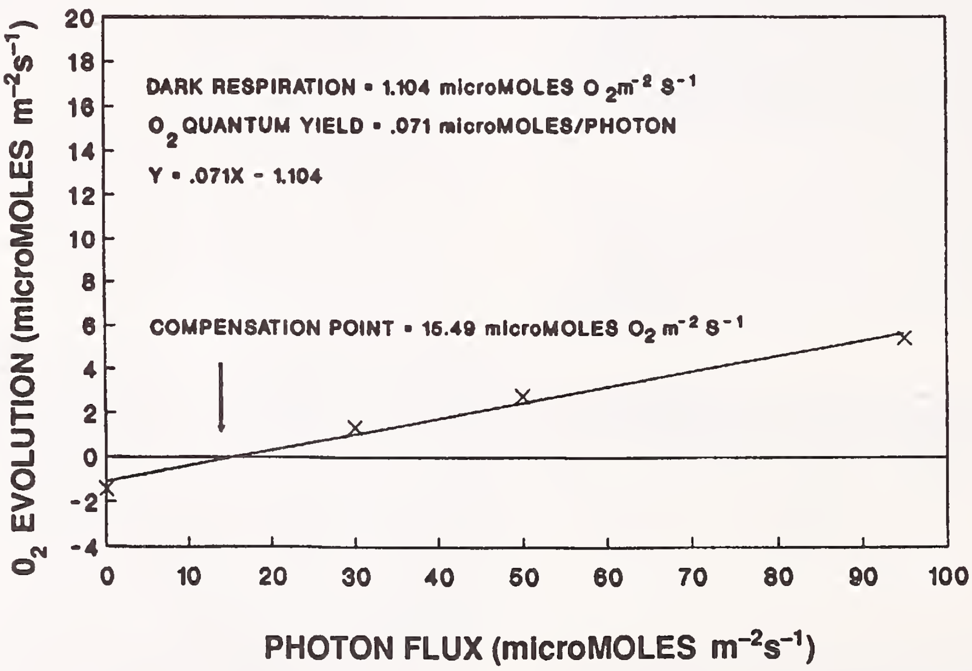

Figure 3—Linear regression on the PFD's up to 95 µmoles/m²/s calculates the light compensation point, quantum yield, and dark respiration. This graph shows the results for tree 123.

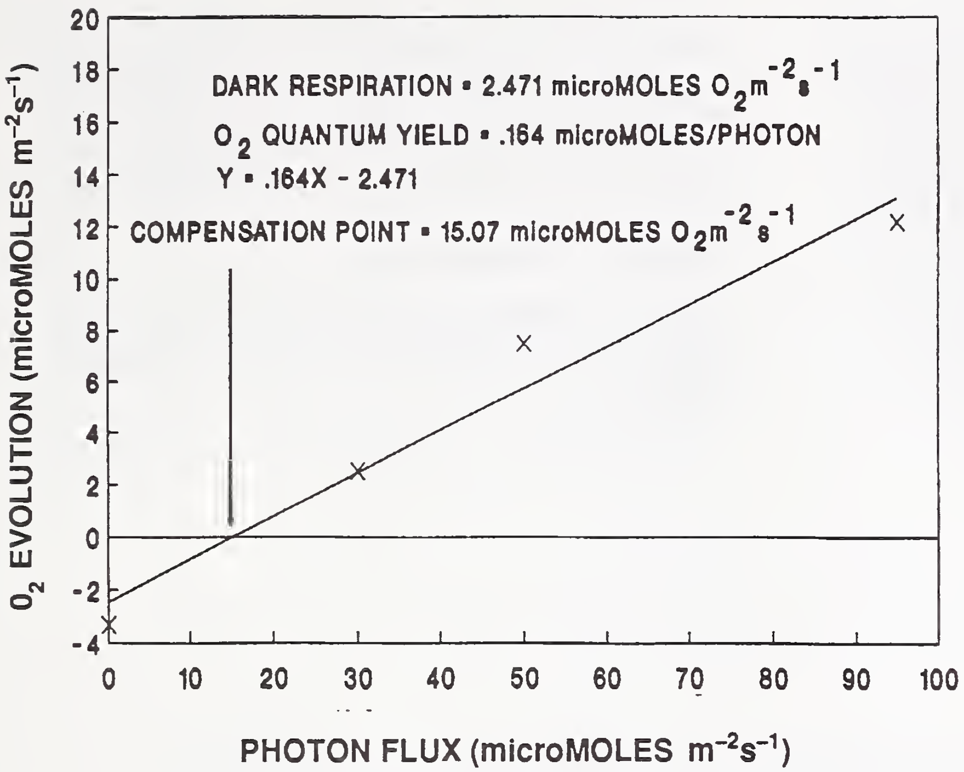

Figure 4—Linear regression on the PFD's up to 95 µmoles/m²/s calculates the light compensation point, quantum yield, and dark respiration. This graph shows the results for tree 144.

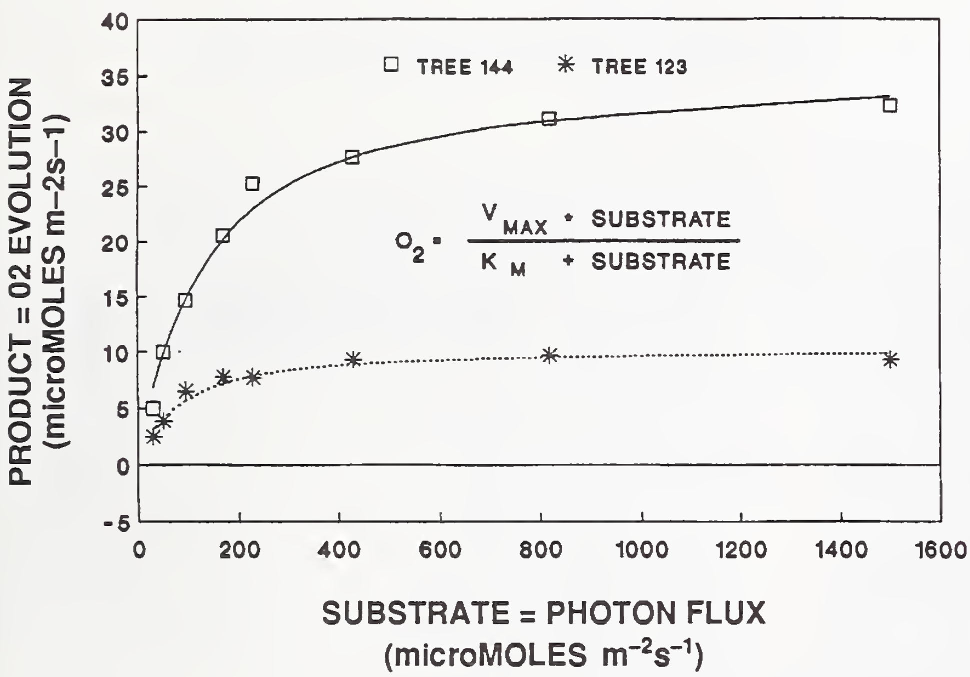

Figure 5—Results of nonlinear regression used to calculate V_{max} and K_m for the Michaelis-Menton enzyme kinetics model. Values used here represent net photosynthesis + dark respiration.

The anaylsis provides estimates of V_{max} (= P_{max} in the assimilation model) and K_m (= Michaelis-Menton constant for light, the light intensity at ½ P_{max}).

Parameters derived from the laboratory measurement of O_2 evolution—maximum photosynthesis (PMAX), the Michaelis-Menton constant for light (Km), dark respiration, and the light compensation point (fig. 6)—were used to calibrate the photosynthesis assimilation model used in Forest-BGC. To calibrate the Forest-BGC assimilation equation, modifications were made as described elsewhere (McDonald these proceedings). The equation we used was:

µmoles CO_2/m²/s = (ICC*MCC*MMC*LMCS)/(MCC + MMC*LMCS); µmoles CO_2 = 0.75 O_2 evolved

where:
- ICC = internal CO_2 concentration = 59,259 ubars = 5 percent CO_2 in O_2 measurement cells
- MCC = maximum stomatal conductance = 0.008 m/s (Lohammer and others 1980)
- MMC = maximum mesophyll conductance = PMAX/ICC (Landsberg 1986)
- LMCS = Light-dependent mesophyll conductance scaler = (current light-light compensation point)/(current light + Michaelis-Menton constant).

RESULTS AND DISCUSSION

Results of calibrating the assimilation model with data from the two different Douglas-fir seedlings are shown in figure 7. The calibrated model was then inserted into Forest-BGC. Output from Forest-BGC after calibration for some assumed situations is shown elsewhere (fig. 6 in McDonald these proceedings). Photosynthesis values in this figure are net photosynthetic rates.

Our calibration procedure, using the Hansatech O_2 evolution cells, seems to be capable of providing the necessary parameters for the assimilation portion of Forest-BGC. Lohammer and others' (1980) equation appears to represent the actual data points (fig. 7).

Since Forest-BGC assimilation is computed from this equation, the procedure described has the potential to calibrate the process model for specific species growing on specific sites with a specific growth history. These parameters taken singly or in combination may also provide useful quantitative indicators of system stress.

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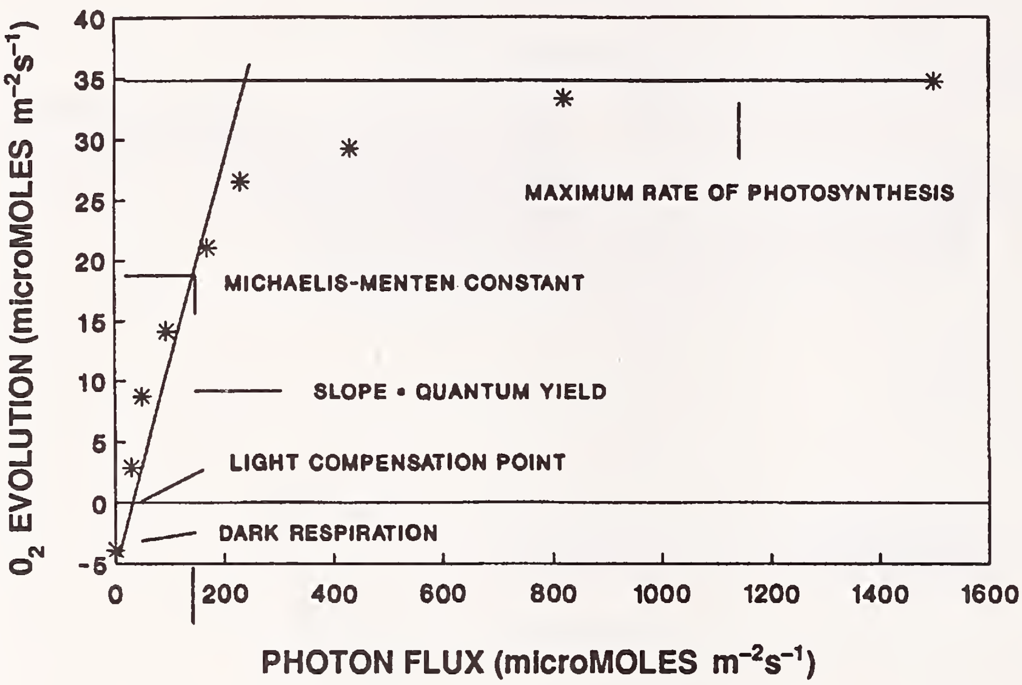

Figure 6—Photosynthetic parameters calculated from the O_2 evolution data using linear and nonlinear regression techniques.

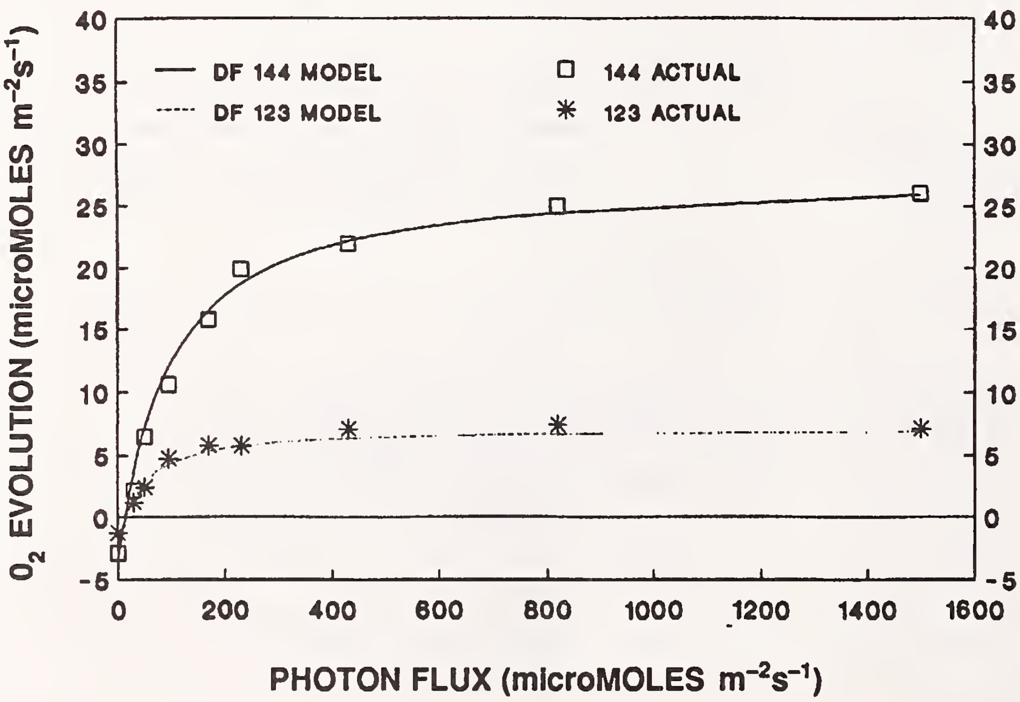

Figure 7—O_2 evolution for trees 123 and 144 plotted on calibrated assimilation model output. Values represent net photosynthesis.

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SOIL ORGANIC MATTER EFFECTS ON DOUGLAS-FIR GROWTH IN NORTHERN IDAHO SOILS

Deborah Page-Dumroese, Russell T. Graham, and Alan E. Harvey

Regeneration of Douglas-fir (*Pseudotsuga menziesii* var. *glauca* [Beissn.] Franco) usually depends on site manipulation and planting to achieve fully stocked stands. Seed that produces superior survival and early growth is a key factor in supplying quality Douglas-fir wood to meet an increased demand for future timber-based products. Planting success in the Intermountain West is hampered by low soil moisture and nutrient stress (Duryea and Lavender 1982). On sites where large amounts of topsoil and organic matter have been removed during the harvest/site preparation sequence, initial tree growth and survival is reduced and continues to be depressed throughout the rotation (Glass 1976; Pehl and Bailey 1978). Douglas-fir can be considered an especially sensitive species in this regard (Graham and others 1989).

Soil, wood, litter, duff, and humus govern the quality of many forest sites in the Inland Northwest because they are an important source of moisture-holding capacity and nutrient storage, which are essential for seedling survival (Harvey and others 1976). Organic matter also provides a favorable microsite for many microbes in the rhizosphere. These microbes can be beneficial or deleterious to tree survival and growth (Harvey 1982).

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NUTRIENT AVAILABILITY

Timber harvesting, site preparation, and slash disposal systems can reduce the nutrient capital of forested ecosystems (Clayton and Kennedy 1985). Nutrient loss is greatest when intensive site preparation or utilization is involved (Jurgensen and others 1990; Leaf 1979). Nutrient deficiencies induced by organic matter removal have been blamed for poor yields and plantation failures on numerous sites (Burger and Kluender 1982; Pritchett 1981; Woods 1981). In the steep, mountainous regions of the Inland Northwest, this loss is often accelerated by erosion and leaching after slash piling, burning, and organic matter removal (Clayton and Kennedy 1985). Site preparation techniques that mound the topsoil and organic horizons together have helped achieve greater conifer survival and growth rates in several instances (Page-Dumroese and others 1987; Shoulders and Terry 1978).

MOISTURE AVAILABILITY

Residual harvesting debris has many physical and chemical properties that make it important to biological processes (Harvey and others 1987). Dead plant materials tie up, then gradually release upon decay, substantial quantities of nutrients and tend to retain more available moisture. Available soil moisture is influenced by soil temperature, moisture content, particle size distribution, evaporative losses, and organic matter content (Cleary 1970). Of these five, organic matter is the most influential, since it represents a major component of soil moisture storage (Brady 1974).

MICROBIAL POPULATIONS

When the forest floor is exposed through harvesting, there is a sharp increase in solar radiation and a concomitant reduction of transpiration. The previously stable microclimate below the organic layer becomes subject to large temperature, moisture, and nutrient fluctuations.

Changes in microsite lead to changes in microbial populations as well. Soil microbes, including mycorrhizae, play a critical role in soil development and plant nutrition and are perhaps an important contributor to seedling success. Harvey and others (1976, 1978, 1987; Harvey 1982) have demonstrated that humus and brown cubical decayed wood are the major substrate for ectomycorrhizal growth on Douglas-fir. These fungi tend to confer drought resistance on seedling roots (Parke and others 1983). Other benefits to the host plant include resistance to pathogens, tolerance to environmental stresses, and an increase in moisture and nutrient uptake (Marx and Krupa 1978). Ectomycorrhizal colonization is highly dependent on inoculum source and environmental conditions surrounding the root system (for example, moisture content, temperature, pH, and organic matter [Slankis, 1974]). Harvest and site preparation alter these factors and may reduce the number of ectomycorrhizae and their diversity (Amaranthus and Perry 1987). It follows, therefore, that soils high in organic matter affect plant rhizospheres through microbial activities and subsequent changes in plant moisture and nutrition.

All these interrelated processes (microbial activity, nutrient and water release) are crucial for the maintenance of conifer plantations in the Inland Northwest and are of particular concern for Douglas-fir. Soil manipulation, in the form of site preparation and harvesting activities, can be a controlling factor for stand establishment or failure.

Because field trials of site preparation techniques have difficulty in separating the influence of organic matter on seedling performance and independent conflicting variables (Graham and others 1989; Page-Dumroese and others 1986, 1989), this study was established.

SITES, MATERIALS, AND METHODS

Soil organic matter, mineral soil, and seed were collected from two dissimilar habitat types in northern Idaho. A low-elevation site (750 m) in the *Thuja plicata/ Clintonia uniflora* (Cooper and others 1987) habitat type and a high-elevation site (3,500 m) in the *Abies lasiocarpa/ Xerophyllum tenax* habitat type (Cooper and others 1987) were selected. Soil was sieved to pass a 2-mm mesh sieve. Planting media consisted of 100 percent organic soil, 100 percent mineral soil, and 50 percent mineral/50 percent organic soils. Soil was tamped into 163-cm^3 cells and seed was sown. Bulk densities of the cells were kept constant (table 1). Seed was sown in April and seedlings were grown until October.

For complete detail of site description, materials, methods, and statistical analyses see Page-Dumroese and others (1990).

Table 1—Average soil bulk density (g/cc), after tamping soil into seedling cells, for each soil mix from each habitat type

Soil mix	THPL/CLUN[1]	ABLA/XETE
	g/cc	
Organic	0.98a	0.98a
50/50 mix	1.00a	1.00a
Mineral	1.11a	1.13a

[1]Habitat types: THPL/CLUN = *Thuja plicata/Clintonia uniflora*, ABLA/ XETE = *Abies lasiocarpa/Xerophyllum tenax*.

RESULTS

Organic matter plays an important role in the growth and performance of Douglas-fir seedlings. Soils containing high organic matter levels had a greater water-holding capacity (Page-Dumroese and others 1986) and, therefore, a greater available water content than soils consisting of only mineral soil. The available water in the organic matter may help bring the seedlings through critical drought periods during the growing season.

Seedlings grown in organic mixes grew significantly better than those grown in only mineral soil (table 2). Height, top weight, root weight, root collar caliper, and terminal bud length were all significantly greater for seedlings grown in the organic mixes. Terminal bud length was used as an indicator of both current season conditions and seedling potential (Kozlowski and others 1973). Seedlings grown in the organic mix soils had a significant growth advantage, not only for the current year, but also for the following season, over their mineral soil counterparts. The improved seedling morphological properties likely can be attributed to the additional moisture and nutrients available in soil organic matter.

The most notable difference between seedlings grown in the various mixes was the number of ectomycorrhizae. Seedlings grown in mineral soil generally had more ectomycorrhizae than those grown in organic mixes. Seedlings grown in the more fertile environments (more moisture and nutrients) may have less need for extensive ectomycorrhizal colonization (Brainerd and Perry 1987). However, seedlings grown in mineral soil did not seem to derive benefit from colonization; neither seedling physical nor chemical properties (tables 2 and 3) were improved. Under this particular circumstance, ectomycorrhizae may represent a carbohydrate cost to seedlings deficient in factors not improved by ectomycorrhizal colonization (Reid 1979).

Seedling chemical properties also reflected the soil type in which they were growing (table 4). Total N and P in seedlings were significantly higher in those grown in 100 percent organic matter than in the 50/50 mix or in mineral soil. This was probably a reflection of the amount of N and P released from the soils during the growing season. Potassium levels did not follow these same trends. This was due to high initial K levels in the mineral soil and subsequent release from exchange sites during the growing season.

Table 2—Physical characteristics of Douglas-fir seedlings grown in three soil mixes

Soil source	Soil mix	Height	Top wt.	Root wt.	Root collar diameter	Bud length	Mycorrhizae per gram of root
		cm	- - - - - - - - *g* - - - - - - - -		- - - - - - - - - *mm* - - - - - - - - -		*No.*
THPL/CLUN[1]	Organic	5.8a	0.14a	0.17a	1.1a	3.6a	0.8b
	50/50	4.8b	.09b	.13a	1.0a	2.9b	4.0ab
	Mineral	4.6b	.06b	.08b	.7b	2.8b	6.5a
ABLA/XETE	Organic	5.4a	.13a	.20a	1.2a	3.0a	2.5a
	50/50	4.4ab	.09ab	.12b	.9ab	2.4ab	3.5a
	Mineral	4.0b	.05b	.07c	.8b	2.2b	4.0a

[1]Habitat types: THPL/CLUN = *Thuja plicata/Clintonia uniflora*, ABLA/XETE = *Abies lasiocarpa/Xerophyllum tenax.*

Table 3—Chemical characteristics of Douglas-fir seedlings grown in three soil mixes

Soil source	Soil mix	Total N	Total P	Potassium
		- - - - *Percent* - - - -		*mg/kg*
THPL/CLUN[1]	Organic	1.96a	0.38a	130.1ab
	50/50	1.33ab	.17b	123.4b
	Mineral	1.17b	.23ab	168.2a
ABLA/XETE	Organic	1.87a	.39a	120.0b
	50/50	1.28b	.13b	173.2a
	Mineral	1.10b	.11b	183.2a

[1]Habitat types: THPL/CLUN = *Thuja plicata/Clintonia uniflora*, ABLA/XETE = *Abies lasiocarpa/Xerophyllum tenax.*

Table 4—Average soil chemical concentrations of each soil mix from each location

Soil source	Soil mix	Total N	Total P	Potassium	pH
		- - - - *Percent* - - - - -		*mg/kg*	
THPL/CLUN[1]	Organic	0.03a	0.27a	5.1	5.7a
	50/50	.03a	.24a	3.3a	6.0a
	Mineral	.01a	.22a	2.4a	6.1a
ABLA/XETE	Organic	.10a	.11a	3.0a	5.4a
	50/50	.03b	.12b	2.6a	5.4a
	Mineral	.03b	.07a	2.5a	5.5a

[1]Habitat types: THPL/CLUN = *Thuja plicata/Clintonia uniflora*, ABLA/XETE = *Abies lasiocarpa/Xerophyllum tenax.*

CONCLUSIONS

Increased nutrient and moisture availability associated with organic soil components, although in many cases not sufficient to be statistically significant in this study, likely resulted in increased uptake and provided the basis for increased seedling growth. This is a strong indication that organic residues left on a site after harvesting could determine the performance of succeeding stands by providing critical moisture and nutrients, especially on harsh (dry) sites low in N and P.

In the Inland Northwest most harvested areas are being planted to achieve adequate stocking levels. This provides an opportunity for land managers to minimize soil horizon destruction and enhance seedling growth through organic matter conservation. Along with maintaining organic matter levels, careful consideration of microsite/seedling relationships is also very likely to improve stand conditions throughout the rotation.

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SOIL PROPERTIES AS A FACTOR IN REGENERATION FAILURES IN THE GRAND FIR MOSAIC—PRELIMINARY RESULTS

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The grand fir mosaic (GFM) consists of a patchwork arrangement of several forest plant communities located in an elevation range of 4,000 to 5,800 feet above sea level. The grand fir mosaic is found primarily in Idaho's Clearwater and Nez Perce National Forests. The plant communities (habitat type from Cooper and others 1987) are:

1. Forested stands where the grand fir/wild ginger (*Abies grandis / Asarum caudatum*) habitat type predominates.
2. Bracken fern (*Pteridium aquilinum*) glades that appear to be climax communities.
3. Alder glades where the Sitka alder/broadleaved montia (*Alnus sinuata / Montia cordifolia*) habitat type predominates.

As the forested areas are harvested, bracken glade communities can invade and exclude most woody vegetation, including conifers. This occurs despite attempts at natural and artificial regeneration. The pocket gopher (*Thomomys talpoides*) also invades the harvested areas and rapidly multiplies. Current research indicates bracken fern allelopathy and damage by pocket gophers as major factors in the poor regeneration success (Ferguson and Boyd 1988). The alder glades expand slightly, causing few problems.

The bracken fern produces a large amount of biomass, which dies in the fall and is buried by snow in the winter. When spring arrives, a surface of almost bare soil is revealed. Thus rapid decomposition and mineralization rates may be taking place beneath the snow.

This study compares forest and natural bracken glade soils. It also examines soils from cutover areas where bracken fern has invaded to see if soil properties are changing.

METHODS AND MATERIALS

The study area is located about 15 miles northeast of Headquarters, ID, in the North Fork Ranger District of the Clearwater National Forest. Portions of the area's forests were clearcut about 25 years ago. Within the study area there is uncut forest, natural bracken fern glades, and harvested areas invaded by bracken fern.

A randomized block design was used for sampling. Soil cores were taken from the uncut forest, bracken fern glades, and the bracken-invaded areas. To reduce variability, the cores were taken from areas that had similar relief, parent material, and climate. Fifteen samples from each area were taken to a 1-foot depth. Soil analysis was completed at the University of Idaho's Analytical Services Laboratory and Soil Fertility Laboratory.

RESULTS

Clearwater National Forest soil chemical properties are given in table 1; physical properties are not yet characterized. More thorough soil sampling, in addition to foliar sampling, is planned for 1990. Statistical analysis as well as finalized results will be presented after these data have been received.

DISCUSSION

The soil chemical properties of the uncut forest and the bracken glade appear to be somewhat different. Soil chemical properties of the bracken-invaded areas differ from both the uncut forest and the bracken glade. However, they seem most similar to bracken glade soils.

The most striking change in the bracken glade-invaded areas is the decrease in pH and increase in extractable aluminum. Within the bracken glades and bracken-invaded areas there are high amounts of composition, mineralization, and nitrification during the winter and early spring. During this time there is little active uptake by plants, and nitrate and cations can be leached from the soil. This process lowers the pH, which could cause aluminum to be released. Bracken fern is also high in phenolic acids, which may aid acidification. Apparently the soil's buffering capacity is not able to neutralize the increased acidity.

The increased amount of decomposition and mineralization could be due to several factors:

1. The bracken glade community produces a greater annual turnover of biomass than the forest community it replaced.
2. A change in microclimate where lack of shade in early spring and late fall causes soil to be warmed to higher temperatures.
3. Warmer and wetter environments than normal for these elevations.

Preventive measures can be based on an understanding of the mechanism that causes this process. In addition, a search for methods of prevention without awareness of the mechanism must continue. Different types of silvicultural practices need to be examined to determine which are least likely to lead to this process.

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Table 1—Clearwater National Forest soil chemical properties—means (standard deviations)

Property, units, method	Sample area		
	Uncut forest	Bracken invaded	Bracken glade
Phosphorus ug/g NaOAC	1.56 (0.75)	2.95 (1.50)	4.25 (1.93)
Potassium ug/g NaOAC	209 (57)	214 (73)	502 (188)
CEC meq/100 g NH4OAC	24.9 (3.8)	30.2 (6.2)	31.6 (3.8)
Exch. Ca meq/100 g NH4OAC	5.6 (1.9)	5.0 (1.5)	3.9 (1.8)
Exch. Mg meq/100 g NH4OAC	0.80 (0.19)	1.12 (0.35)	1.21 (0.46)
Exch. K meq/100 g NH4OAC	0.80 (0.18)	1.15 (0.32)	1.75 (0.51)
Zinc ug/g DTPA	0.73 (0.67)	1.66 (1.14)	2.05 (1.32)
Manganese ug/g DTPA	45.4 (17.3)	28.6 (15.7)	32.0 (14.2)
Copper ug/g DTPA	0.32 (0.09	0.36 (0.15)	0.41 (0.13)
Extrac. Al ug/g KCL	35 (43)	197 (128)	211 (101)
pH pH units 1:1 water	5.93 (0.26)	4.95 (0.38)	4.79 (0.30)
Carbon percent CHN analyzer	4.77 (0.95)	5.18 (1.79)	6.14 (1.53)
Nitrogen percent CHN analyzer	0.28 (0.05)	0.41 (0.12)	0.45 (0.09)

CONCLUSIONS

1. The different plant communities lead to formation of different soils on what otherwise appear to be areas with the same relief, climate, and parent material.
2. When bracken fern invades harvested areas, soil chemical properties can be altered. Thus, in addition to allelopathy and pocket gophers, a change in soil properties may be contributing to the poor regeneration success.

A NOTE ON THE MOSAIC

The introduction stated that the grand fir mosaic (GFM) exists in both the Clearwater and Nez Perce National Forests. However, as more information is gathered, it seems that the GFM is not the same ecosystem in each Forest. For instance:

1. The "climax" bracken fern glades are almost nonexistent in the Nez Perce National Forest.
2. After harvest, bracken fern is the primary invader in the Clearwater National Forest, but in the Nez Perce National Forest the primary invader is coneflower (*Rudbeckia occidentalis*).
3. Soil chemical properties have not changed after harvest in the Nez Perce National Forest.

Because of this, this poster paper was largely focused on one National Forest. The Clearwater was chosen because it is more interesting from a soils standpoint. However, from a forestry standpoint, both Forests are important because regeneration failures in the GFM have been noted in each.

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